

# Control of Agricultural Nonpoint Source Pollution in Kranji Catchment, Singapore

by

Margaret A. Hoff

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Signature of Author \_\_\_\_\_  
Margaret A. Hoff  
Department of Civil and Environmental Engineering  
May 10, 2013

Certified by \_\_\_\_\_  
Peter Shanahan  
Senior Lecturer of Civil and Environmental Engineering  
Thesis Supervisor

Accepted by \_\_\_\_\_  
Heidi M. Nepf  
Chair, Departmental Committee for Graduate Students



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## Abstract

Singapore's Kranji Reservoir is highly sensitive to nutrient and bacterial pollution, both of which can be directly traced to agricultural runoff. Water quality samples were collected along the main drainage channel in the Neo Tiew subcatchment, which drains to Kranji Reservoir, in an effort to determine the source and degree of agricultural nonpoint source pollution in the area. Grab samples collected from eight sampling locations along the reach of the drainage channel under wet- and dry-weather conditions were analyzed for nitrogen, phosphorus, and bacterial species, as well as total suspended solids. High nutrient and bacterial concentrations were observed at sampling locations in the upstream region of the subcatchment, with total nitrogen as high as 19.8 mg/L, total phosphorus as high as 2.12 mg/L, and a peak total coliform count over 1,000,000 MPN/100 mL. The peak concentration of most of the observed contaminants occurred directly downstream from an intensive row-cropping vegetable production operation. These observations indicate that this farming operation is a primary, though not sole, contributor to nonpoint source pollution in the area. A constructed free-water-surface treatment wetland was designed to treat runoff immediately downstream from the identified source. The designed wetland is projected to remove, depending on flow conditions, between 13 and 99% of influent total phosphorus, 51 to 99% of influent total nitrogen, greater than 99% of influent fecal coliform, and approximately 75% of influent total suspended solids. Agricultural management practices for mitigating runoff contamination are also recommended, including cyclic irrigation and crop rearrangement. It is evident that agricultural nonpoint source pollution is a significant water quality concern in the Neo Tiew subcatchment in particular and the Kranji catchment in general, but there are a number of promising and practical options to address this problem.

Thesis Supervisor: Peter Shanahan

Title: Senior Lecturer of Civil and Environmental Engineering



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# 1. Introduction

## 1.1 Singapore Water Management

This section was written collaboratively with Ndeye Awa Diagne.

Singapore is lauded by many, including the World Health Organization, as an archetype of an integrated water resources management model (Chen et al. 2011). This recognition is not because the small city-state has abundant water. On the contrary, it lacks sufficient naturally occurring water resources to sustain its population of 4.8 million. Water limitations are serious enough to warrant Singapore's inclusion by the United Nations on its list of water-scarce countries (Ong 2010). Though the average annual rainfall of 2,400 mm is above the global average, the country lacks the land area necessary to harvest an adequate amount of that precipitation (Tan et al. 2009). Furthermore, the small island has no other sources of renewable freshwater, lacking the volumes of surface- and groundwater that typically sustain other countries. The thirsty country, which currently consumes approximately 1.36 billion liters of water per day (Tortajada 2006), is projected to continue to grow, reaching a population of 6.5 million in the next 50 years (Chen et al. 2011), further stressing its already scarce water resources.

Singapore scores remarkably high on a measure of the proportion of the population with adequate water supply, both in terms of quantity and quality. As Chen et al. (2011) report, 100% of the population has consistent access to water of sufficient quantity to meet their consumption demands. Furthermore, 99.96% or higher of that water supply meets the World Health Organization (WHO) drinking water standard, which, though not a universal standard, is generally considered sufficient to ensure potability of water. Similarly, 100% of the population is reported to have access to "adequate sanitation" (Chen et al. 2011). The country's impressive performance in light of water scarcity is the result of careful management by Singapore's national water utility, the Public Utilities Board (PUB), of the country's four "National Taps," its four sources of water (Tan et al. 2009).

As previously mentioned, Singapore gets an above-average amount of rainfall, but the country simply is not physically large enough to collect enough of that water as it falls. This spatial limitation has long been the target of engineering projects in Singapore and has resulted in an intricate network of rainwater collection channels and reservoirs, considered the country's first National Tap (Tan et al. 2009). The rainwater collection system provides about 50% (Chen et al. 2011) of Singapore's daily water consumption of 1.36 billion liters (Tortajada 2006). Efforts to expand the ability to harvest precipitation are continuing, including progressive rooftop harvesting schemes and continuous expansion of the reservoir network, with the aim of transforming 90% of Singapore's land area into water catchment. Despite the advanced technology and the government's aggressive expansion of rainwater collection systems, physical limitations still necessitate other sources of water to meet the country's needs (Tan et al. 2009).

Singapore's second National Tap is imported water from Johor, Malaysia, which makes up another 40% of its water supply (Chen et al. 2011). Singapore has imported a large percentage of

its water since it separated from the Federation of Malaysian States in 1965, but the relationship has often been tense and uncertain in the intervening years. At various times, Malaysia has threatened to cut off the water supply for political or economic reasons and agreement on pricing has been a long-standing issue (Chen et al. 2011). There is currently an agreement in place that will provide water to Singapore through 2061 at a price of less than S\$0.01 per 1,000 liters, but further terms are uncertain (Tortajada 2006). Driven by at-times acrimonious relations with Malaysia, Singapore has investigated other international sources for water, including Indonesia, but has been deterred by high development costs and the inherent insecurity of relying on other nations for natural resources (Chen et al. 2011). Most recently, Singapore has invested significant financial and political resources into careful water resource management, as well as the development of its third and fourth National Taps, desalination of seawater and reuse of wastewater, with the ultimate goal of national water independence (Tortajada 2006).

The country's first large desalination plant, the Tuas Desalination Plant, opened in 2005 with a price tag of S\$200 million (Chen et al. 2011). Though desalination technology is improving rapidly, it still has relatively low capacity and is highly energy intensive. Accordingly, the Tuas plant can supply 113 million liters per day (less than 7% of the country's current water demand) at a cost of S\$0.78 per 1,000 liters (Tortajada 2006). For a sense of the economics, this water source is more than seventy times more expensive than imported water, but, as of 2011, was still the lowest cost seawater desalination plant in the world (Chen et al. 2011). High costs and lagging technology in desalination have encouraged Singapore to explore water reuse technology, which typically has lower economic costs than desalination but higher social barriers.

Reuse of highly treated wastewater has been explored as an alternative water source in Singapore since 1972, with the first operational treatment plant built in 2000 (Tortajada 2006). The recycled waste stream and fourth National Tap, locally termed "NEWater," is currently produced at four facilities across the country and will ultimately account for more than 30% of the national water supply (Chen et al. 2011). Though treated to a higher level than necessary to meet standards for human consumption, the majority of NEWater is currently used for industrial water needs rather than domestic (potable) distribution. Since 2003, a small percentage of the recycled water has been designated for indirect potable use, in which the highly treated effluent is mixed into existing raw water sources (Ching 2010). The percentage of NEWater designated for indirect potable use is expected to rise, but will still remain much lower than industrial usages (Tortajada 2006). Just as with desalination, production costs will likely drop as the technology evolves, but current reuse treatment costs are already low at approximately S\$0.30 per 1,000 liters, which is less than half the cost of desalination (Tortajada 2006).

Singapore's success in water provision, particularly in the arena of water reuse, has largely been attributed to the organization of its water management institution, the Public Utilities Board. Since 2001, PUB has managed the entire water cycle within the country, including potable water delivery, sewage, waste treatment, and rainwater collection (Tan et al. 2009). In addition to controlling the entire water cycle, PUB was also given general autonomy over its functions, which has allowed the agency unilateral authority over all aspects of water governance, including pricing structures, regulatory frameworks, and enforcement mechanisms (Tortajada 2006). This structure is touted by many to "eliminate administrative barriers in water management and make implementation effective and efficient" (Chen et al. 2011). Furthermore, PUB is widely

considered to effectively include the private sector when appropriate and foster public acceptance and political will through its success (Tortajada 2006).

In 2006, PUB launched the Active Beautiful Clean Waters (ABC Waters) Programme, a strategic initiative to open Singapore's reservoirs and waterways to the public for recreational activities. The larger objective of the ABC Waters Programme is to make Singaporeans cherish their water bodies and be more conscious of water scarcity (PUB 2009). The recreational activities include kayaking, fishing, barbecue, and picnic activities, and may involve direct contact with the water bodies. However, water quality of the reservoirs and waterways has been a concern for PUB. In fact, recent studies have reported contamination in the nation's water reservoirs and the stormwater drains feeding them. Both urban and agricultural runoff has been reported to contain high levels of pollutants including suspended solids, nutrients, heavy metals, and pathogenic bacteria (Wang 2012). In order to protect public health, there have been ongoing studies and investigations to evaluate the levels of contamination within reservoir catchments and bacteria loading to the reservoirs and waterways (Chua et al. 2010).

## **1.2 Kranji Reservoir Water Quality**

Kranji Reservoir is a large drinking water reservoir in the northwestern region of Singapore and is surrounded by a wide variety of land uses, dominated by a mixture of agricultural activities. Historically, Kranji has been plagued by the same water quality issues as the rest of Singapore's reservoirs and has had particularly high concentrations of nutrients and bacteria (NTU 2008). Agricultural nonpoint source pollution has been identified as a major contributor to the nutrient and bacteria contamination in Kranji Reservoir (NTU 2008). An investigation of storm-event-mean concentrations at a variety of locations in the Kranji watershed (Figure 1) has revealed a significant contribution from a sampling location in the Neo Tiew subcatchment (Le 2013). Specifically, in a study of wet-weather nonpoint-source runoff from around the Kranji catchment, samples collected in the Neo Tiew subcatchment at a location draining two plant nurseries and one vegetable farm (Figure 2) presented maximum event-mean concentrations of nitrogen, phosphorus, and total suspended solids (TSS). In the study, which included composite samples from 10 storm events in 2011, average event-mean concentration of total nitrogen at the Neo Tiew sampling point was 13.2 mg/L, total phosphorus was 3.2 mg/L, and TSS was 522 mg/L (Le 2013). These concentrations, as well as those of constituent chemical species of nitrogen and phosphorus, were the highest observed around the Kranji catchment by Le (2013).





Figure 1: Map of Kranji catchment and subcatchments (NTU 2008)



Figure 2: Neo Tiew subcatchment sampling location (Le 2013)



A 2008 study of water quality in the Kranji Reservoir by Nanyang Technological University evaluated bacteriological contamination at a sampling point in the Neo Tiew subcatchment near Le's Neo Tiew sampling location (NTU 2008). The mean cell density of *E. coli* of 14 samples collected at the location was 4,560 Most Probable Number (MPN)/100 mL. The mean enterococci density was similarly high at 2,410 MPN/100 mL (NTU 2008). Additionally, past studies by Master of Engineering students from the Massachusetts Institute of Technology (MIT) have observed high levels of bacterial contamination in agricultural runoff in the Kranji catchment. *E. coli* density as high as 290,000 colony-forming units (CFU)/100mL was found in nonpoint source runoff in the Neo Tiew subcatchment (Dixon et al. 2009). Bossis (2011) observed very high concentrations of biological contamination indicators in agricultural runoff in the Kranji catchment. The highest observed values were 21,000,000 MPN/100 mL of fecal coliform (FC), 10,000,000 MPN/100 mL *E. coli*, and 370,000 MPN/100 mL enterococci, all of which were observed in areas draining plant nurseries. Though the samples were taken in a different subcatchment, the measured concentrations indicate the significant bacterial contamination that can be present in agricultural runoff (Bossis 2011).

Nutrient and bacteria contamination in Kranji is of concern for a number of reasons. First, Kranji is a phosphorus-limited environment in which inputs of phosphorus have been shown to cause blue-green algae blooms. This eutrophication, in addition to creating objectionable odor and aesthetic conditions, can lead to hypoxia of the reservoir environment upon decomposition (NTU 2008). Though nitrogen is not the limiting nutrient in the Kranji environment, water samples from the reservoir with elevated nitrogen levels have been shown to stimulate increased chlorophyll production relative to samples without elevated nitrogen inputs (NTU 2008). Lastly, the presence of total coliform, *E. coli*, and enterococci in surface water is indicative of human fecal bacteriological contamination. Human contact with microbial contamination in water can result in intestinal illness and death (Douglas-Mankin and Okoren 2011).

It is clear that in order for Kranji to meet the goals of the ABC Waters Programme, nitrogen, phosphorus, and bacterial contamination of the reservoir must be mitigated to the extent possible. As the agricultural operations in the Neo Tiew subcatchment have been identified as substantial contributors of these contaminants, a mitigation plan needs to be developed for this nonpoint source of pollution. Before a remedy can be prescribed, a determination must be made as to the primary source of these contaminants. The agricultural activities in the Neo Tiew subcatchment include containerized plant nurseries, organic fruit and vegetable production, and intensive row crop vegetable production. Additionally, there are non-agricultural activities in the subcatchment, including large-scale sand storage for concrete production, construction equipment storage, and military training. Once it is determined which land uses are driving the high pollutant concentrations in the Neo Tiew drainage channel, a solution can be devised. This study endeavors to determine the source of the bacteria and nutrient contamination observed in the Neo Tiew subcatchment and to propose pollution control solutions. The pollution control plan includes upstream source mitigation strategies and design of a downstream contaminant remediation system.

## 2. Nonpoint Source Pollution

Nonpoint source pollution is an increasing problem in surface waters around the world, with contributions from agriculture of particular concern. Different agricultural activities result in different contaminant profiles in runoff, but the primary constituent contaminants and treatment options share similarities across different agricultural industries (Novotny and Olem 1994). Most relevant for this study are explorations of runoff from plant nursery operations, intensive row crop farms, and organic produce cultivation, but studies of remediation of other high-intensity agricultural and landscaping activities with high fertilizer or nutrient application rates are applicable as well. There have been several studies aimed at understanding and categorizing the environmental consequences of runoff from the highly intensive types of agriculture employed in the Neo Tiew subcatchment. Additionally, some studies have built on that understanding, or similar studies regarding other types of agriculture, and explored Best Management Practices, or BMPs, that can be used to remediate contaminated agricultural runoff. Studies of BMPs can largely be divided into those addressing structural BMPs, such as treatment wetlands and vegetative filter strips, and those addressing non-structural BMPs, such as irrigation management and crop selection. The body of literature is largely applicable to the study of agricultural runoff in the Kranji Reservoir, but there remain some gaps in the academic knowledge.

### 2.1 Agricultural nonpoint source pollution

He et al. (2006) studied phosphorus concentrations in runoff from both citrus groves and vegetable farms in Florida under climatic and topographic conditions that closely resemble those of Singapore. The conditions described on the vegetable farms in the study are similar to those on the vegetable farms in the Neo Tiew subcatchment, particularly the prevalence of artificial drainage channels and relatively flat landscape. He et al. (2006) found wide variation in total phosphorus concentrations, ranging from 0.01 to 22.74 mg/L, with a concentration greater than 1 mg/L in a majority of the observed samples. Furthermore, there was generally a greater proportion of dissolved phosphorus, the most bioavailable form of phosphorus (Logan 1993), than of particulate phosphorus (He et al. 2006).

Udawatta et al. (2004) also endeavored to understand the contribution to phosphorus contamination of runoff from agricultural fields, but under different conditions than He et al. (2006). Udawatta et al. (2004) studied runoff from a subcatchment dominated by commercial-scale row crop farming of corn and soybeans in the central United States. Though different in scale and climate, similarities still exist between the massive row crop agriculture in the United States and the intensive form of it practiced in Singapore, particularly the exposure of bare soil to precipitation and irrigation, which results in greater losses of sediment in runoff (Udawatta et al. 2004). The study determined that the concentration of phosphorus in a given sample of agricultural runoff was influenced by the runoff volume, soil type, concentration of phosphorus in the soil, precipitation, and antecedent soil moisture conditions. Over the six-year study period, the mean total phosphorus concentration in runoff from the subcatchment was 0.87 mg/L (Udawatta et al. 2004).

The Neo Tiew subcatchment also includes an organic fruit and vegetable farm. Though organic farms do not apply chemical or synthetic fertilizers, they often apply nutrients to crops in other

forms. There are many forms of biological fertilizers applied on organic farms, including animal manure and silage, both of which can result in agricultural runoff with high concentrations of nutrients and bacteria (Brown 1993). As Brown (1993) reports, “badly managed organic farms are potentially as polluting as an over-fertilized conventional farm.” Due to the very broad nature of the term “organic” and the multitude of agricultural activities that it encompasses, a reliable comparison of runoff from organic farming to runoff from conventional farming is difficult, but studies indicate that runoff from organic facilities contains less nitrate than that from conventional farms (Brown 1993).

Given the presence of multiple nurseries in the Neo Tiew subcatchment, as well as the similarity between the intensive type of vegetable cultivation practiced in the area to plant nursery practices, a foundational survey of typical containerized nursery runoff and pollution control is useful for understanding how the runoff from Neo Tiew compares to typical nursery wastewater. Alexander’s (1993) examination of large nursery operations in the United States provides an overview of typical nursery practices in irrigation, fertilization, and pollution control. It is apparent from the study that many nurseries over-irrigate, resulting in leaching of nutrients and bacteria from fertilizer and increased volumes of wastewater (Alexander 1993).

Craig and Weiss (1993) endeavor to apply the USDA Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) model to the overland and subsurface transport, degradation, and uptake of pesticide runoff from nurseries. The study provides an overview of common pesticides and indicates that the quantity and quality of runoff is largely dependent upon site-specific factors such as soil content, field slope, pesticide application rates, and types of pesticides applied as well as the characteristics of a given rainfall event. While surface water contamination by pesticide runoff is undeniably an important topic, the complexity represented by the diverse chemistry of pesticides places this issue outside the scope of this study.

A survey of the literature on runoff from the types of agriculture within the Neo Tiew subcatchment indicates that the nutrient pollution profiles previously observed at Neo Tiew are not atypical. Nurseries and vegetable farms often practice high fertilizer loading rates and frequent irrigation in an attempt to accelerate plant growth, resulting in leaching of nutrients from soil and large volumes of runoff. Alexander (1993) found a wide range of nitrate-N concentrations in nursery runoff, ranging from 1.6 to 55.0 mg/L. Total phosphorus concentrations were also widely distributed, ranging between 0.01 and 1.95 mg/L. Variations were found to be related to time elapsed since fertilizer application and rainfall rate (Alexander 1993). Taylor et al. (2006) observed significant seasonal variability in nitrate-N concentrations in runoff from a nursery in Georgia, a state with a relatively temperate climate. Nitrate-N concentrations ranged from 11.1 to 29.9 mg/L in spring months and 2.8 to 5.2 mg/L in winter months. Phosphorus concentrations did not significantly vary seasonally and exhibited a range of 0.7 to 2.2 mg/L phosphate-P (Taylor et al. 2006). Due to the relatively stable annual climate in Singapore, seasonal variations are not likely to be observed in the Neo Tiew runoff. Huett et al. (2005) studied nutrient content of nursery runoff and observed a mean annual nitrate-N concentration of 9 mg/L and phosphate-P concentration of 2 mg/L. Sharma et al. (2008) found a very large range of nitrogen and phosphorus concentrations at nurseries in Florida, with nitrate-N ranging from 0.2 mg/L to 300 mg/L and phosphate-P concentrations between 0.5 and 144 mg/L. These observed typical values are summarized in Table 1 with the comparable values from Le’s (2013) study of the Neo Tiew sampling location.

**Table 1: Typical values of nitrogen and phosphorus concentrations in agricultural runoff.**

<b>Study</b>	<b>Type of agriculture</b>	<b>Nitrogen mg/L nitrate-N</b>	<b>Phosphorus mg/L phosphate-P</b>
He et al. 2006	Vegetable row crop	n/a	0.79
Udawatta et al. 2004	Commodity row crop	n/a	0.87 <sup>a</sup>
Alexander 1993	Plant nursery	1.6-55	0.01-1.95 <sup>a</sup>
Taylor et al. 2006	Plant nursery	2.8-29.9	0.7-2.2
Huett et al. 2005	Plant nursery	9	2
Sharma et al. 2008	Plant nursery	0.2-300	0.5-144
Le 2013	Mixed (Neo Tiew)	8	1.2

a: Total phosphorus values reported in lieu of phosphate-P

As can be seen from Table 1, there is wide variability among studies of agricultural runoff, likely due to the differences in plant type, geographical and climatic variations, rainfall events, and fertilizer application. Despite the large range of observed data, it is clear that the average event-mean concentrations observed at the Neo Tiew sampling location are well within the typical range for runoff from intensive agriculture. Though the losses of nitrogen and phosphorus observed in agricultural runoff may be slight as compared to the mass of fertilizer applied, the typical observed concentrations could very easily result in significant changes to an aquatic ecosystem (He et al. 2006). While many studies mention the occurrence of bacterial contamination in addition to nutrient loading in nursery runoff (Alexander 1993, EPA 1992), none provide a quantified range for typical bacterial loading values. Given the broad variability of observed runoff concentrations, it is more useful to discuss the nutrient and bacterial reduction performance of BMPs in terms of percentage reductions rather than absolute numerical concentration reduction values.

## **2.2 Nutrient reduction targets**

In order to design effective source management programs and treatment systems, it is helpful to have a reduction target in mind. Nutrient and bacteria concentration limits for surface water are not well defined in the United States, or in Singapore, as the broad variety of aquatic environments and land uses makes standardization extremely difficult. Some of the many factors affecting a surface water body's nutrient capacity include its area, depth, hydraulic residence time, seasonal variability, turbidity, and watershed factors (Gibson et al. 2000). Despite these complications, attempts have been made at setting limits for nutrient loads in surface water, specifically the United States Environmental Protection Agency's (EPA) proposed criteria for nitrogen and phosphorus in Florida water bodies (White et al. 2011). Even this proposed regulation did not set specific numerical criteria but rather deferred to field assessments of nutrient assimilation capacity of water bodies (White et al. 2011). The EPA recommends not exceeding total nitrogen of 0.5 mg/L and total phosphorus of 0.1 mg/L in surface waters, but this is a recommendation and not a regulatory limit (Sharma et al. 2008, Taylor et al. 2006). Currently, limits exist for nitrate-N in drinking water (10 mg/L) due to its association with blue baby syndrome, but concentration limits do not exist for recreational surface water bodies (Douglas-Mankin and Okoren 2011).

Pesticide use and runoff is currently regulated in the United States, and many agricultural operations are aware that similar regulations may soon be enacted for nutrients as well (White et

al. 2009). One possible manifestation of such regulations may be a Total Maximum Daily Load (TMDL), in which a daily maximum quantity of discharge of a given pollutant is set for each individual source and impaired receiving-water body, but the process of establishing these limits is particularly complex for nonpoint sources such as plant nurseries and vegetable farms (Taylor et al. 2006).

Bacterial concentration limits in surface water are better defined than those for nutrients. In the US, fecal coliform limits in surface water vary based on the water body's intended usage, categorized by drinking, primary contact (swimming waters), and secondary contact (boating and fishing waters). The limit for primary contact water is 200 CFU/100 mL and the limit for secondary contact water is 2,000 CFU/100 mL (Douglas-Mankin and Okoren 2011). This limit applies for the water body as a whole, but a loading limit for individual nonpoint sources does not exist. The U.S. EPA recommends monitoring *E. coli* and enterococci as indicator bacteria in recreational waters, rather than total coliform, as total coliform can be from a number of sources while *E. coli* and enterococci are specifically fecal in origin (EPA 2012). Though a numerical pollution reduction target may be impossible to specify, the previously observed high concentration levels from the Neo Tiew drainage channel indicate that any possible reductions will be worthwhile.

### **3. Sample collection and analysis**

Field work for this project was conducted in January 2013. Water quality samples were manually obtained throughout the Neo Tiew subcatchment under both dry and wet weather conditions. Sampling points were selected at strategic locations along the primary drainage channel in the Neo Tiew subcatchment in an effort to determine the primary source of nutrient and bacterial contamination. Instantaneous measurements of flow depth and channel dimensions were similarly observed during both dry and wet weather. All laboratory analyses of collected samples were also performed in Singapore in January 2013.

#### **3.1 Procedures**

##### **3.1.1 Site selection**

Sampling sites were selected on the bases of drainage area land usage and sampling feasibility. Some sections of the drainage channel were inaccessible for sampling purposes due to either restricted property access or vegetation overgrowth. Restricted properties include the military training grounds and the government sand storage area (see Figure 3). Access to the drainage channel from Green Circle Farm and the downstream construction and storage property was inhibited by excessive vegetation. The remaining accessible sample sites were selected in an effort to represent the diversity of land uses along the drainage channel and to facilitate isolation of particular areas of concern. Six sampling sites were initially selected and numbered from upstream to downstream locations. Additional sampling locations were added throughout the field work period to aid in elucidating sources of nutrient and bacterial contamination. A map of the sampling locations can be seen in Figure 3.

Following are brief descriptions of each sampling location.

Location 1: Samples were collected directly from the primary outfall at Mao Sheng Quanji Landscaping nursery. This outfall is the first point of discharge into the drainage channel. A photograph of this location can be seen in Figure 4.

Location 2: Samples were collected from the most upstream point of the drainage channel on Farm 85 Trading property. This point is the convergence of the lots of Green Circle Farm, Kiat Lee Landscape, and Farm 85 Trading. A small cascade was observed within the channel at the border between the property lots. Samples were collected from this cascade. This location can be seen in Figure 5.

Location 3: This location is the most downstream point of the channel on Farm 85 Trading property, immediately upstream of the sand storage area. This location is depicted in Figure 6.

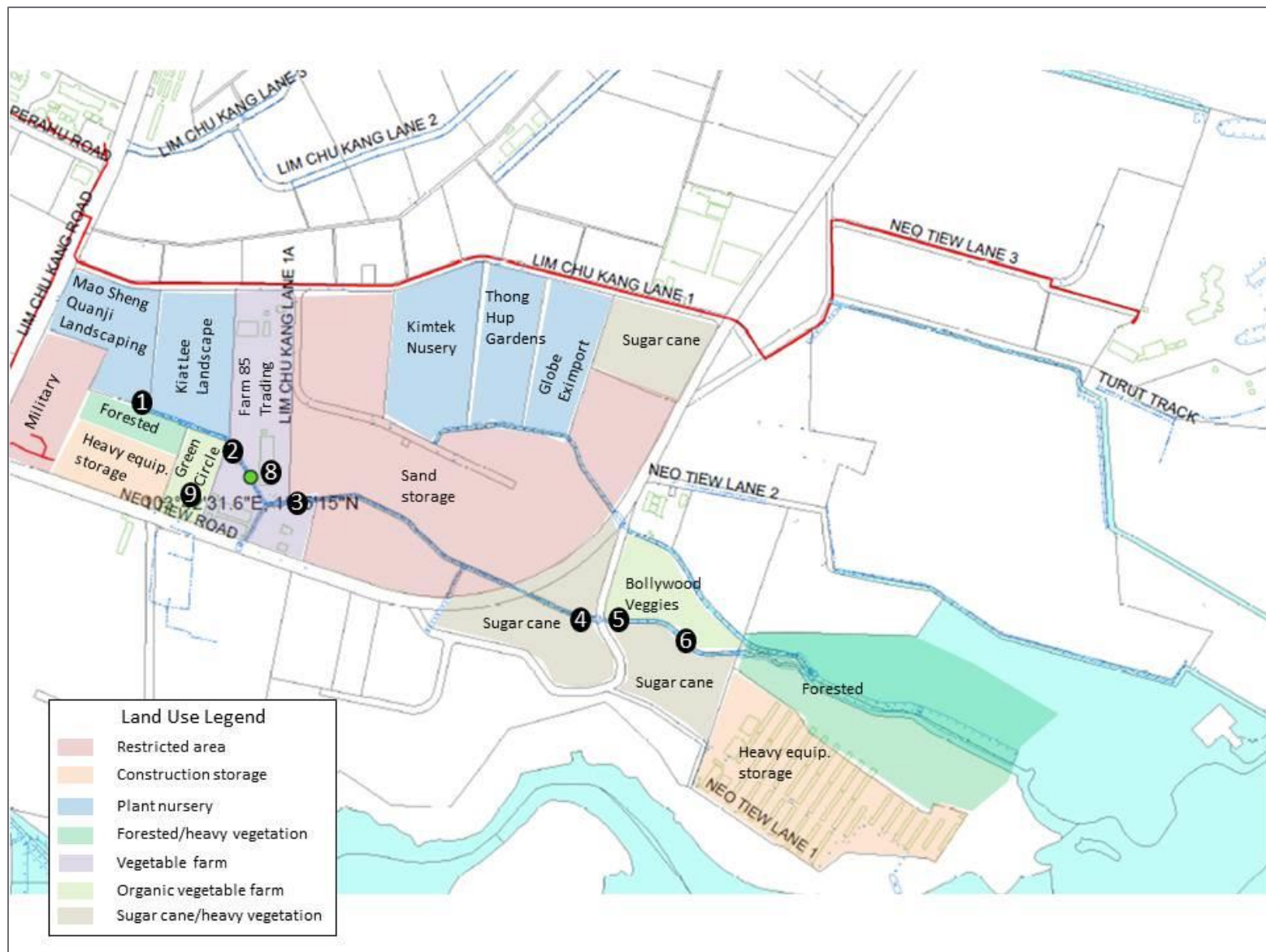


Figure 3: Map of land use and sampling locations in Neo Tiew subcatchment.





**Figure 4: Sampling location 1.**



**Figure 5: Sampling location 2.**





**Figure 6: Sampling location 3.**

Location 4: This sampling location is located on the upstream side of the Neo Tiew Road bridge over the drainage channel. This location is the closest accessible point to the downstream edge of the sand storage property. The sampling location can be seen in Figure 7.

Location 5: Samples were collected immediately downstream of the Neo Tiew Road bridge. Though physically proximate to Location 4, an outfall draining the adjacent sugar cane fields enters the drainage channel between the two sampling points. This sampling location is presented in Figure 8.

Location 6: This location is the downstream side of the pedestrian bridge adjacent to Bollywood Farm. This location was selected in an effort to assess the nutrient and bacteria contributions from Bollywood Farm. The location is depicted in Figure 9.

Location 7: Location 7 was used as a placeholder for field blanks. Blank sample collection methods are described in the following section.

Location 8: Samples were collected from a parallel drainage channel on Farm 85 Trading property. The location was approximately midway between sampling locations 2 and 3 and solely includes drainage from Farm 85 Trading. An image of the sampling location can be seen in Figure 10.

Location 9: Samples were collected from the rainwater collection pond at Green Circle Farm, which is an organic fruit and vegetable farm. Farm personnel indicated that the pond is approximately 6 m deep, but samples were only taken near the water surface. There is a thick layer of vegetation on the top of the pond, as can be seen in Figure 11.

Geographic coordinates of each sampling location and additional location information can be found in Appendix A.



**Figure 7: Sampling location 4.**



**Figure 8: Sampling location 5.**



**Figure 9: Sampling location 6.**



**Figure 10: Sampling location 8.**





**Figure 11: Sampling location 9.**

### **3.1.2 Weather conditions**

Samples were collected under both dry and wet weather conditions in order to provide as complete an understanding of catchment runoff water quality as possible. Rainfall was measured with a RIMCO 8020 tipping bucket rain gauge (McVan Instruments Pty. Ltd, Scoresby, Victoria, Australia) with 0.2 mm tip. The rain gauge was installed for the monitoring and sampling of stormwater from the Kranji Catchment under a joint project between Nanyang Technological University and PUB (Le 2013).

On the first day of sampling, January 16, 2013, there was no rain throughout the morning, during which all samples were collected. The most recent rainfall event recorded at the nearby Verde rain gauge station prior to January 16 sampling ended approximately 18 hours before sampling began and totaled 11.8 mm over its duration of approximately one hour (Le 2013).

There was light rain during sampling on January 22, 2013. The rain event began 3 hours before the start of sampling and ceased shortly after sampling began. Total precipitation from the event, as measured at the Verde station, was 1.2 mm (Le 2013).

### **3.1.3 Sample collection**

All water samples were manually collected as grab samples during January field collection. Water was collected from the drainage channel either directly in laboratory glassware and Nasco Whirl-Pak® (Nasco, Fort Atkinson, WI) collection bags or, in sampling locations in which direct manual physical to the channel was not feasible due to topography or vegetation, using a 10-liter bucket and rope. Equipment used for collection by bucket can be seen in Figure 12. The bucket was rinsed with water from each sample location prior to collecting the sample volume. Immediately following collection by the bucket method, water samples were transferred to laboratory glassware and Whirl-Pak® bags. Samples taken at each location consisted of one liter collected in a brown glass laboratory bottle, two 530-mL Whirl-Pak® bags, and one brown glass 40-mL vial. The large glass bottles were prepared by Setsco Services Pte. for sample collection

and thus were not rinsed or otherwise altered prior to on-site sample collection. Sterile Whirl-Pak® bags were unsealed immediately prior to sample collection in each location. The 40-mL glass vials were rinsed thoroughly with sample water prior to being filled.



**Figure 12: Bucket collection equipment at sampling location 6.**

Upon sample collection, the 1-L bottles and Whirl-Pak® bags were stored on ice for field preservation and subsequently transferred to a 4°C refrigerator until laboratory analysis was completed. All holding time practices were conducted in accordance with EPA standards (EPA 1979). The 40-mL glass vial sample from each location was collected for ammonia nitrogen (NH-N) analysis and thus was preserved accordingly until time of analysis. For adequate preservation, 0.05 mL of concentrated sulfuric acid ( $\text{H}_2\text{SO}_4$ ) was added in the field to each 40-mL sample in accordance with EPA standard method 350.1 (EPA 1979). Following the addition of sulfuric acid, samples were stored on ice until analysis was completed.

A field blank was prepared and transported on each day of sampling in order to provide insight into the reliability of the sample collection and analysis process. The blank consisted of deionized water collected directly into Whirl-Pak® bags and laboratory glassware immediately prior to field collection of samples. The blank samples were transported on ice during sample collection and refrigerated identically to field samples. Analysis of the blank sample was performed in accordance with the same standards and procedures as those applied to field samples.

#### **3.1.4 Flow determination**

In order to estimate flow volume and thus pollutant loads, channel dimensions were measured at locations of supercritical flow under both dry and wet conditions. Sampling locations 2 and 8 were such areas of supercritical flow, both of which can be approximately represented as geometric channels with critical flow for the purpose of flow determination. Channel dimensions and depth of flow were then used in conjunction with the appropriate hydraulic equations to determine flow rates at these sampling locations.

Wet conditions were observed on the morning of January 22, 2013 after 3 hours of precipitation totaling 1.2 mm. Dry conditions were observed on the morning of January 25, 2013, with the most recent precipitation event having ceased 17 hours prior to observation (Le 2013).

The supercritical flow region at sampling location 2 can be approximated by a trapezoidal channel, as seen in Figure 13. The respective flow region at sampling location 8 can be represented by a circular channel, as seen in Figure 14.



Figure 13: Supercritical flow region at sampling location 2 with overlaid trapezoid.



Figure 14: Supercritical flow region at sampling location 8 with overlaid arc.

The depth of flow at each of these locations of supercritical flow is approximately equivalent to critical depth, which, along with other relevant dimensions, can be related to channel flow by Equations 1 and 2 (Brater and King 1976):

For flow in trapezoidal channels:

$$Q = \sqrt{g \frac{(b+zD_c)^3}{b+2zD_c}} D_c^{\frac{3}{2}} \quad (1)$$

For flow in circular channels:

$$Q = \frac{2^{\frac{5}{2}} g^{\frac{1}{2}} (\theta - \frac{1}{2} \sin 2\theta)^{\frac{3}{2}}}{8 (\sin \theta)^{\frac{1}{2}} (1 - \cos \theta)^{\frac{5}{2}}} D_c^{\frac{5}{2}} \quad (2)$$

In which:

- $Q$  = channel flow ( $\text{m}^3/\text{s}$ )
- $g$  = acceleration due to gravity ( $\text{m}/\text{s}^2$ )
- $b$  = width of bottom of trapezoidal channel (m)
- $z$  = slope of sides of trapezoidal channel (dimensionless)
- $D_c$  = critical depth (m)
- $\theta$  = angle from center of channel to edge of flow (radians)

Using the above-listed equations and dimensions collected in the field, the following flow rates were calculated (Table 2). Measured channel dimensions are provided in Appendix B.

**Table 2: Calculated channel flow at locations 2 and 8.**

<b>Date</b>	<b>Flow, location 2 (trapezoidal)</b>	<b>Flow, location 8 (circular)</b>
1/22/2013 (wet weather)	350 $\text{m}^3/\text{day}$	69 $\text{m}^3/\text{day}$
1/25/2013 (dry weather)	150 $\text{m}^3/\text{day}$	26 $\text{m}^3/\text{day}$

### 3.1.5 Chemical and biological analysis

The collected samples were analyzed for total nitrogen (TN), ammonia nitrogen (NH-N), nitrate nitrogen and nitrite nitrogen (NO-N), total phosphates (TP), total dissolved phosphorus (TDP), total organic phosphorus (TOP), total suspended solids (TSS), total coliform, *E. coli*, and enterococci. Bacterial analyses (total coliform, *E. coli*, and enterococci) were performed using IDEXX Quanti-Trays and growth media (IDEXX Laboratories, Inc., Westbrook, ME, USA). Analyses of NH-nitrogen, NO-nitrogen, and TP were performed using CHEMetrics Vacu-vials® and photometric analysis (CHEMetrics, Inc., Midland, VA, USA). TSS, TN, TOP, and TDP for all sampling locations, as well as NO-N nitrate nitrogen for selected locations, were analyzed by an external laboratory, Setco Services Pte., according to American Public Health Association (APHA) standard procedures (APHA 2012).

Analyses of NO-nitrogen (both nitrite and nitrate species), NH-nitrogen, and total phosphate were performed on the same day as sampling. Samples taken for bacterial analysis were stored in a refrigerator at a temperature of 4°C for approximately 24 hours prior to serial dilution and incubation. The water samples collected in 1-L glass bottles were delivered to Setco Services Pte. on the same day of sampling for analysis of total suspended solids, total organic phosphorus, total dissolved phosphorus, and total nitrogen.



#### 3.1.5.1.1 *CHEMetrics chemical analysis*

Analysis of nitrate and nitrite nitrogen, ammonia nitrogen, and total phosphates were performed using CHEMetrics Vacu-vials® and colorimetric analysis. The analytical kits for each chemical species were used in accordance with the manufacturer's instructions, which comply with applicable APHA and EPA analytical procedures (CHEMetrics 2012).

Nitrate-N concentration was determined using a reduction of nitrate to nitrite in the presence of zinc and other chemicals provided in the CHEMetrics analysis kit, ultimately resulting in a pink-hued solution, the color of which is proportional to the concentration of nitrate present (CHEMetrics 2012b). Vials of prepared nitrate-N solution can be seen below in Figure 15. A photometer set to the appropriate CHEMetrics analytical program was used to translate the resultant color to the nitrate-N concentration. Samples were prepared and analyzed in accordance with CHEMetrics procedure K-6913, which meets APHA standard 4500-NO<sub>3</sub><sup>-</sup> E and EPA method 353.3 (CHEMetrics 2012b). Samples collected at locations 1, 3, 4, 5, 6, and 8 were over the nitrate-N concentration range quantifiable by the CHEMetrics method, so nitrate-N analysis on these samples was performed by the external analytical laboratory.



Figure 15: Nitrate samples prepared for colorimetric analysis using CHEMetrics method.

The CHEMetrics method of nitrite-N analysis relies on the formation of an azo dye from nitrite in an acidic solution in the presence of organic substances. The manufacturer's procedure, K-7003, is compliant with APHA method 4500-NO<sub>2</sub><sup>-</sup> B and EPA method 354.1 (CHEMetrics 2012c).

CHEMetrics analysis of ammonia-N follows the salicylate method, in which monochloramine reacts with salicylate to produce 5-aminosalicylate, the green color of which is used for photometric ammonia concentration determination. Ammonia sample preparation and analysis was performed in accordance with CHEMetrics procedure K-1403, compliant with EPA method 350.1 (CHEMetrics 2012d).

Total phosphate analysis is performed by an acid digestion process in which phosphates are converted to orthophosphate. Reaction of orthophosphates with chemicals in the CHEMetrics solution results in a blue complex to enable colorimetric analysis. Analysis was performed



following the manufacturer's procedure K-8540, which meets APHA standards 4500-P B.5 and E, as well as EPA methods 365.2 and 365.4 (CHEMetrics 2012e). All analyses were performed on the same day of sampling and analyzed using a CHEMetrics Model V-2000 photometer.

#### **3.1.5.2 Bacterial analysis**

Determination of bacterial contaminant concentration in samples was performed using IDEXX Quanti-Tray®/2000 in conjunction with IDEXX Enterolert and Colilert growth substrates (IDEXX Laboratories, Inc., Westbrook, Maine). In this method, a 100-mL volume of water sample is combined with the appropriate substrate and deposited in a calibrated, sterilized tray and subsequently incubated to enable appropriate bacterial growth. The results of this process are used to determine the Most Probable Number of colony-forming bacteria per 100 mL (MPN/100 mL) (IDEXX 2012). Separate samples must be prepared for analysis of enterococci using Enterolert and *E. coli* using Colilert. Sample trays prepared using Colilert serve a dual purpose, resulting in counts of both total coliform and *E. coli* (IDEXX 2012).

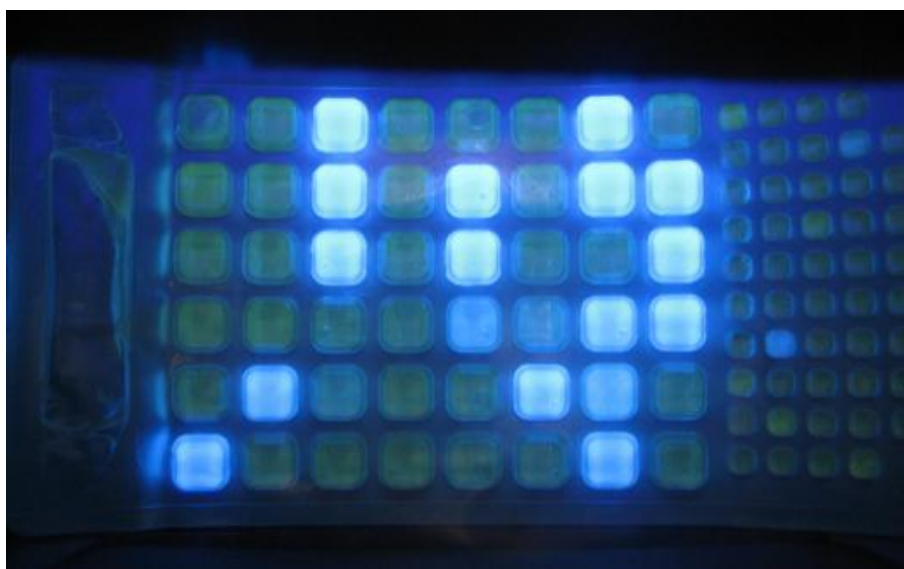
Quanti-Tray/2000 methods facilitate determination of bacterial concentrations ranging from 0 to 2,419.6 MPN/100 mL (IDEXX 2012). Since agricultural samples from the Neo Tiew subcatchment had been previously demonstrated to sustain bacterial concentrations greater than this quantifiable range (Bossis 2011, Dixon et al. 2009), serial dilutions were used to increase the quantifiable range of the analytical method. In the first set of collected samples, water samples from locations 1 and 2 were diluted and prepared at ratios of 1, 1:100, and 1:10,000. Samples collected from all other sampling locations, and all samples from the second day of field collection, were diluted and prepared at ratios of 1:10, 1:1,000, and 1:100,000 due to high anticipated bacterial concentrations. All dilutions were prepared with sterilized glassware and deionized water in order to minimize potential sources of laboratory contamination.

Enterolert and Colilert growth substrates were then added to each dilution, resulting in six total samples from each sampling location. Each sample and substrate mixture was then deposited in a Quanti-Tray/2000, sealed, and incubated for 24 to 28 hours. Enterococci samples were incubated at 41°C, and total coliform/*E. coli* samples were incubated at 35°C, in accordance with manufacturer specifications (IDEXX 2012).

After adequate incubation, concentration of the respective types of bacteria is determined by counting the number of positive and negative wells in each Quanti-Tray/2000. In the trays using Colilert substrate, wells that are positive for total coliform are yellow in color, as can be seen in Figure 16. Under a 6-watt, 365-nm ultraviolet light, wells on the same tray that are positive for *E. coli* fluoresce to a yellow shade. When trays using Enterolert growth media are viewed under ultraviolet light, wells positive for Enterococci fluoresce to a blue shade (IDEXX 2012), as can be seen in Figure 17.



**Figure 16: Total coliform presence in Quanti-Tray/2000. Yellow wells are positive for total coliform.**



**Figure 17: Enterococci presence in Quanti-Tray/2000. Blue fluorescent wells are positive for enterococci.**

Counts of positive and negative wells in each tray are used with the IDEXX Most Probable Number table to determine the MPN for each bacterial type at each dilution. Given that each water sample has three dilutions, there are three possible concentrations to use as the appropriate MPN for each sample, at least one of which is within the range of the upper and lower detection limits. In addition to the MPN for a given positive well count, IDEXX provides a statistical range for each positive count. Thus, the selected value of MPN for each sample is the dilution for which the statistical range is smallest. MPN and statistical range values were multiplied by the appropriate dilution factor to account for the adjusted concentrations.

### 3.1.5.3 External laboratory

Certain chemical analyses were outside the scope of the limited laboratory resources available during the field work period, and thus were performed by a third-party analytical laboratory, Setsco Services Pte., Singapore. Samples to be analyzed by Setsco were collected in glassware provided by the laboratory, preserved on ice during field collection, and delivered to the laboratory within four hours of sample collection.

Setsco performed analyses of total suspended solids, total nitrogen, total organic phosphates, and total dissolved phosphates for all sampling locations, as well as nitrate nitrogen on samples from the locations that were outside the CHEMetrics quantifiable range.

External laboratory analyses were performed in accordance with the following APHA standard procedures (APHA 2012): TSS – APHA: Pt 2540D; TN – APHA: Pt 4500-N org; OP – APHA: Pt 4500-P (G); TDP – APHA: Pt 4500-P (H), and nitrate – APHA: Pt 4500-NO<sub>3</sub> (I).

## 3.2 Results

The full set of measured concentrations can be seen in Appendix C.

### 3.2.1 Nutrients

A broad range of nutrient concentrations were observed in sampling locations along the main drainage channel. The minimum and maximum values observed for each nutrient contaminant on each sampling day can be seen in Table 3.

Table 3: Range of observed nutrient concentrations.

Nutrient	1/16/13		1/22/13	
	Min (mg/L)	Max (mg/L)	Min (mg/L)	Max (mg/L)
Total N	1.8	19.8	4.03	18.1
Nitrate-N	0.3	16.8	1.07	17.0
Nitrite-N	0.08	0.23	0.09	0.24
Ammonia-N	0.04	2.14	0.52	2.66
TP	0.52	2.12	0.27	1.6
TDP	0.16	1.27	0.12	1.2
TOP	0.034	1.04	0.006	0.22

It is clear from the results in Table 3 that nutrient concentrations were not steady throughout the reach of the drainage channel but rather varied by location, so it is important to understand where the maxima and minima occurred. The observed nutrient concentrations at each sampling location, plotted from the most upstream sampling location to the most downstream sampling location, are presented in Figures 18, 19, 20, and 21. As can be seen in Figures 18 and 19, the maximum observed values of total nitrogen and nitrate nitrogen occurred at location 3 on both sampling days. The magnitude of the maximum observed concentrations of TN and NO<sub>3</sub>-N were similar across the two sampling days as well, with TN values of 19.8 and 18.1 mg/L on 1/16/13 and 1/22/13, respectively, and NO<sub>3</sub>-N values of 16.8 and 17 mg/L on 1/16/13 and 1/22/13, respectively. Similarly, maxima of all phosphorus species on both sampling days occurred at

sampling location 3. There was greater variation in maximum phosphorus species concentrations across the two days of sampling, particularly in total organic phosphorus, with a value of 1.04 mg/L on 1/16/13 and 0.22 mg/L on 1/22/13. Though sampling location 3 presented maximum concentrations for most of the nutrient constituents, peak ammonia nitrogen was observed at sampling location 2 on both sampling days. The maximum observed NH<sub>4</sub>-N values were similar across both days of sampling, with values of 2.14 and 2.66 mg/L on 1/16/13 and 1/22/13, respectively.

Though there is some variation across nutrient species and sampling days, a general trend is evident in Figures 18, 19, 20, and 21. There is no strong trend between sampling locations 1 and 2, with some contaminants decreasing, others increasing, and still others holding steady. However, with the exception of ammonia nitrogen and nitrite nitrogen, a clear trend emerges at location 3. At location 3, most nutrient species significantly increase, and then subsequently decrease at location 4, remaining fairly steady through locations 5 and 6. Ammonia nitrogen exhibits a similar trend, except with the strong peak at location 2. Nitrite nitrogen does not exhibit any strong peak, but rather maintains a fairly constant and low magnitude across all sampling locations.

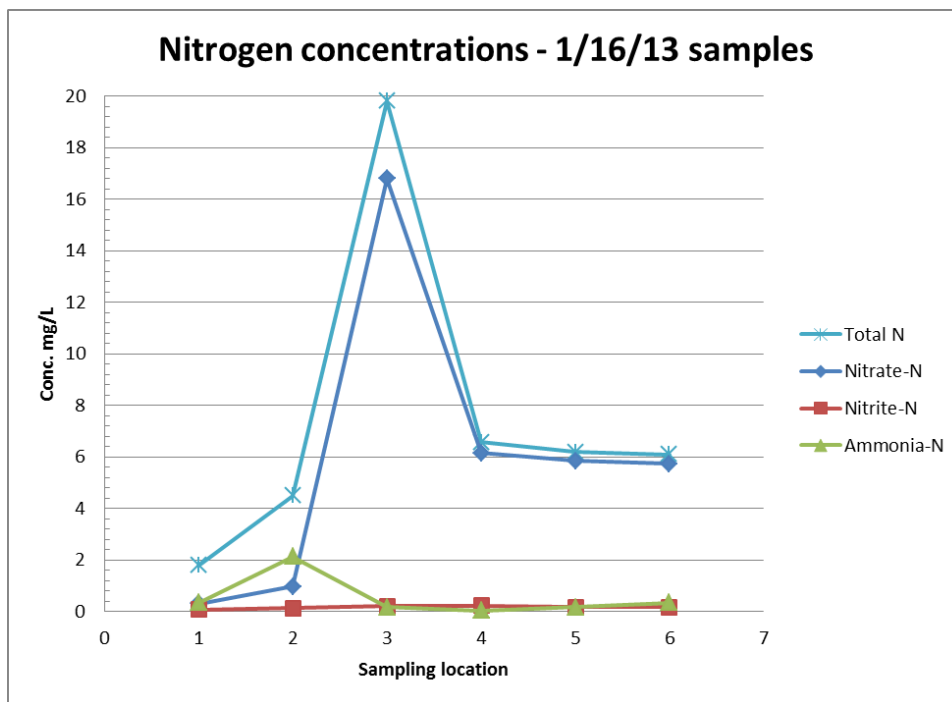


Figure 18: Nitrogen concentrations on 1/16/13.

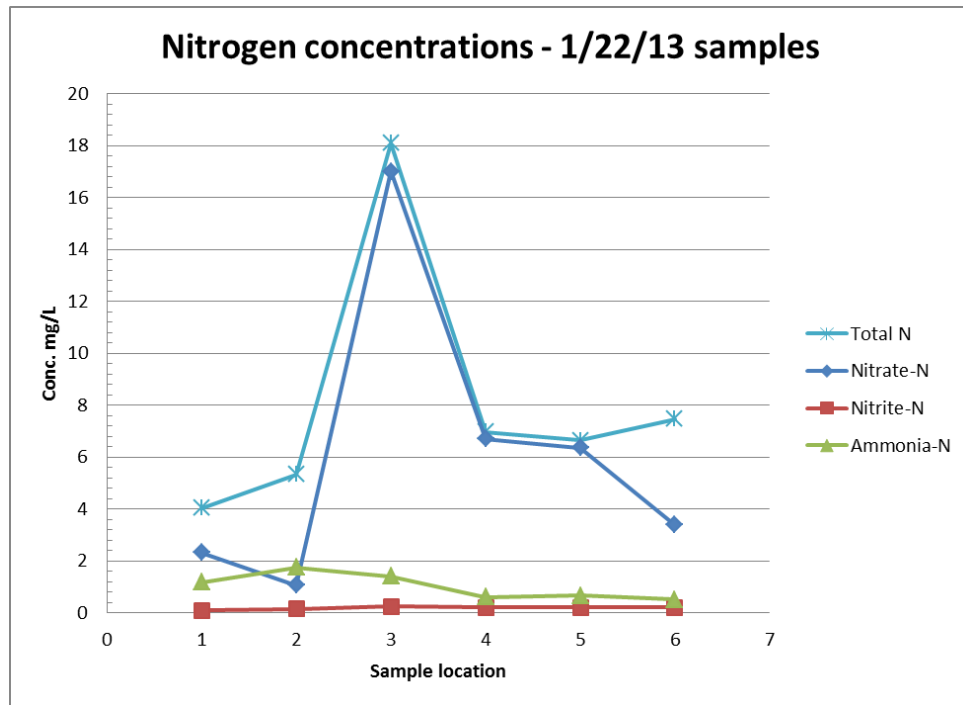


Figure 19: Nitrogen concentrations on 1/22/13.

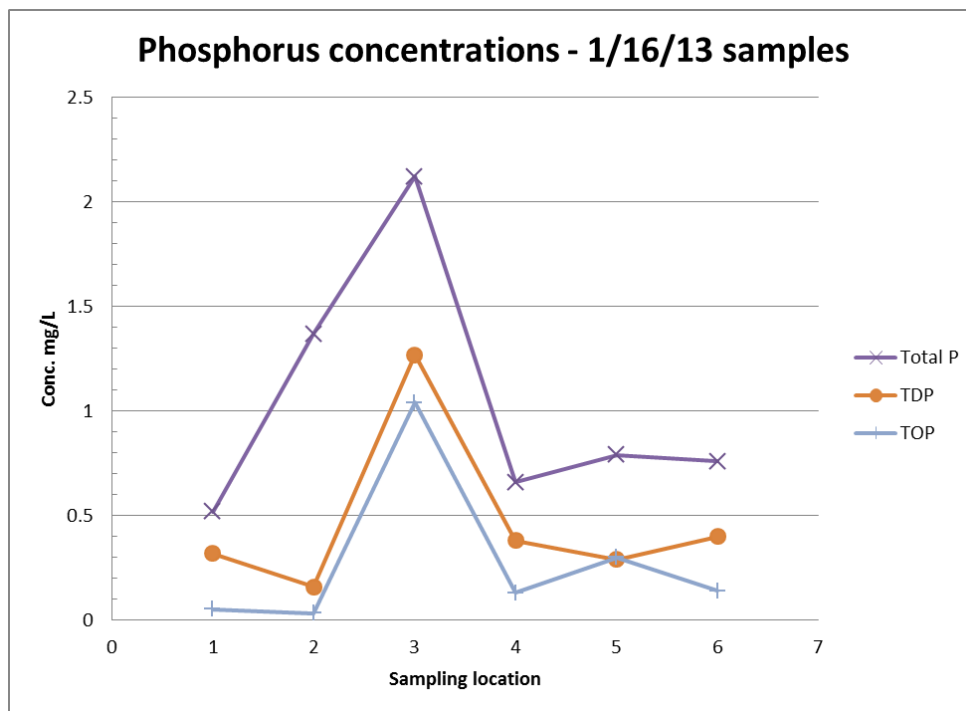
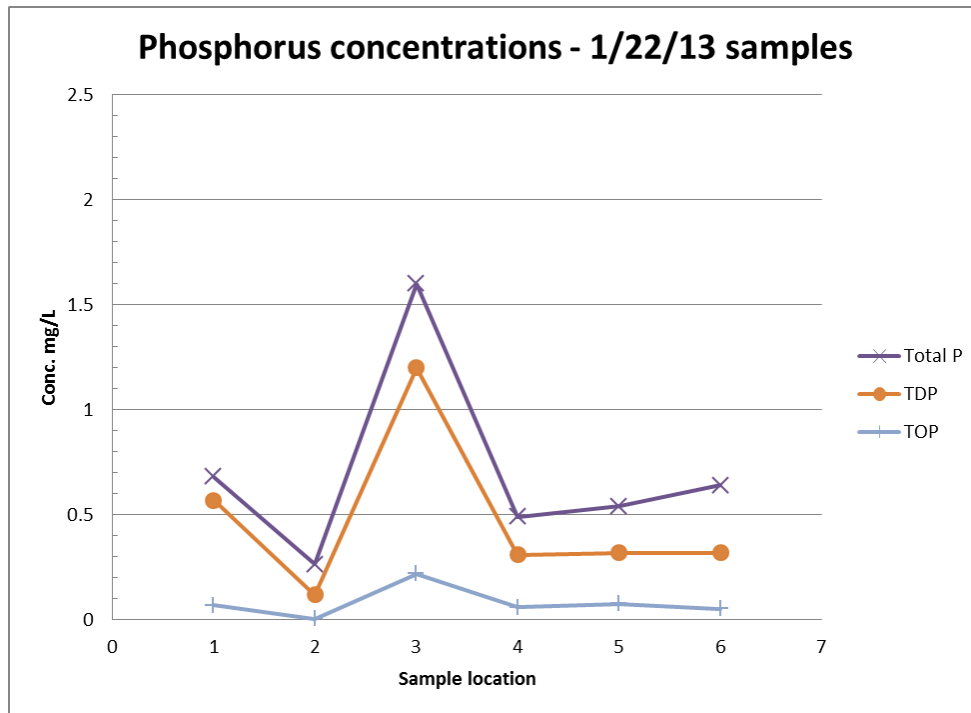


Figure 20: Phosphorus concentrations on 1/16/13.



**Figure 21: Phosphorus concentrations on 1/22/13.**

In addition to the samples collected along the reach of the main drainage channel, the additional locations sampled present interesting trends in nutrient concentration. These additional sampling locations, referred to previously as locations 8 and 9, have been inserted into the graphs of nutrient concentrations along the reach of the drainage channel in order to elucidate the approximate contribution of each of the drainage areas that they represent, as can be seen in Figures 22, 23, 24, and 25. Location 8, the parallel drainage channel on the Farm 85 Trading plot, has been inserted between locations 2 and 3 to represent the contributions from the activities at Farm 85 Trading, and location 9, the pond at Green Circle Farm, has been inserted between locations 1 and 2, though Green Circle Farm is not the sole contributor to runoff between locations 1 and 2.

As can be seen in Figures 22 and 23, the parallel drainage channel at Farm 85 Trading presents much higher concentrations of total nitrogen and nitrate nitrogen than in the main drainage channel at location 3, with values of 81.3 mg/L total nitrogen and 78.8 mg/L nitrate nitrogen on 1/16/13 and 55 mg/L total nitrogen and 53.2 mg/L nitrate nitrogen on 1/22/13. Though the flow in the parallel channel is less than that of the main channel and thus a comparison of the concentrations at locations 8 and 3 is not a direct one, the high total nitrogen and nitrate nitrogen concentrations in the parallel channel may help to explain the high concentrations observed at location 3.

Similarly, the highest observed ammonia nitrogen value was in the pond at Green Circle Farm (location 9) on 1/22/13, with a value of 2.77 mg/L. This overall maximum value immediately precedes the maximum value observed in the main channel at location 2 and can help to explain the trend observed for ammonia in the main drainage channel. The maximum value of total dissolved phosphorus concentration was also observed in the Green Circle pond, 2.07 mg/L,

which was nearly twice the highest observed in any location in the main drainage channel. There was also a spike in total phosphate concentration in the Green Circle pond, though it did not exceed the maximum value observed on that day of sampling, which occurred at location 3.

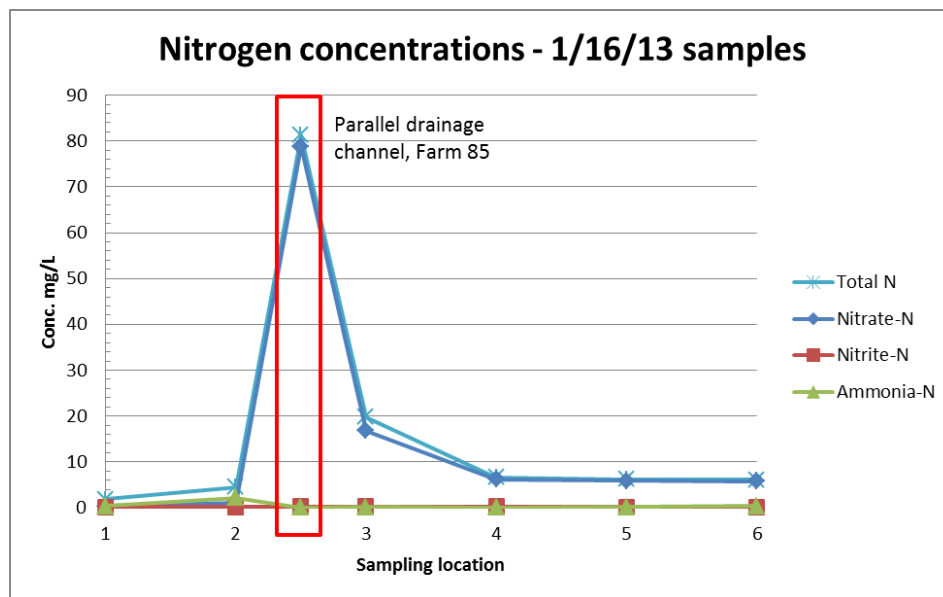


Figure 22: Nitrogen concentrations with additional points, 1/16/13.

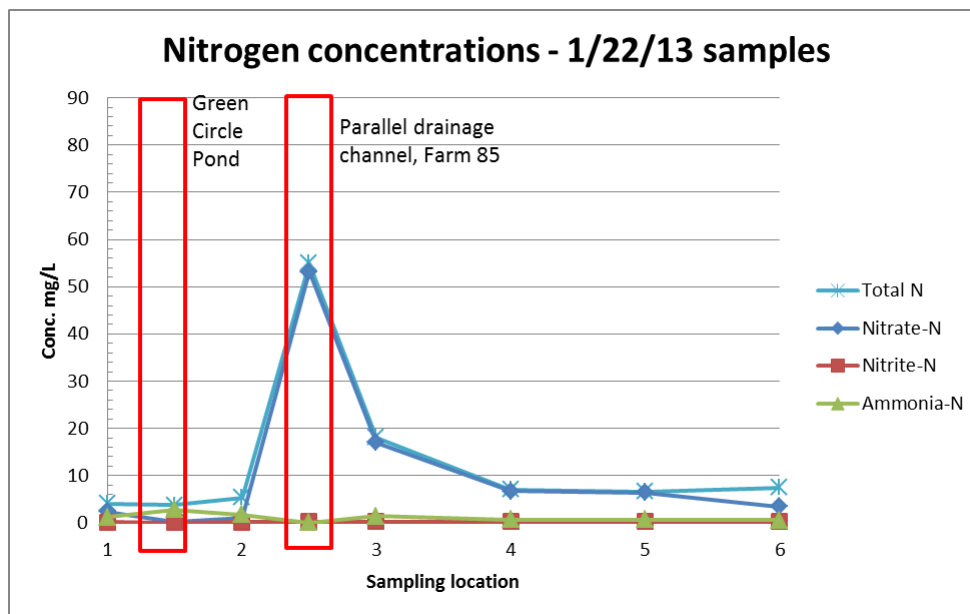


Figure 23: Nitrogen concentrations with additional points, 1/22/13.

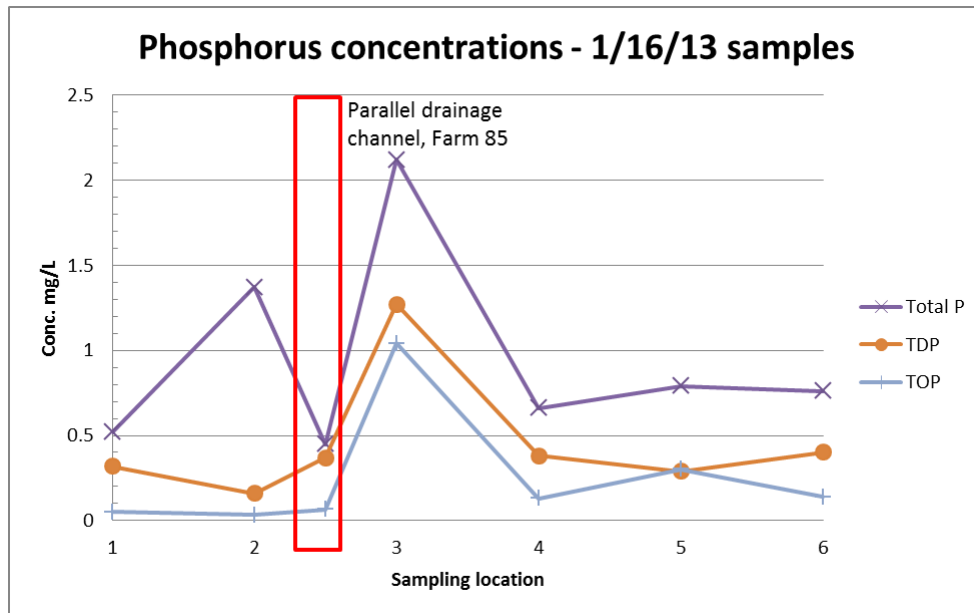


Figure 24: Phosphorus concentrations with additional points, 1/16/13.

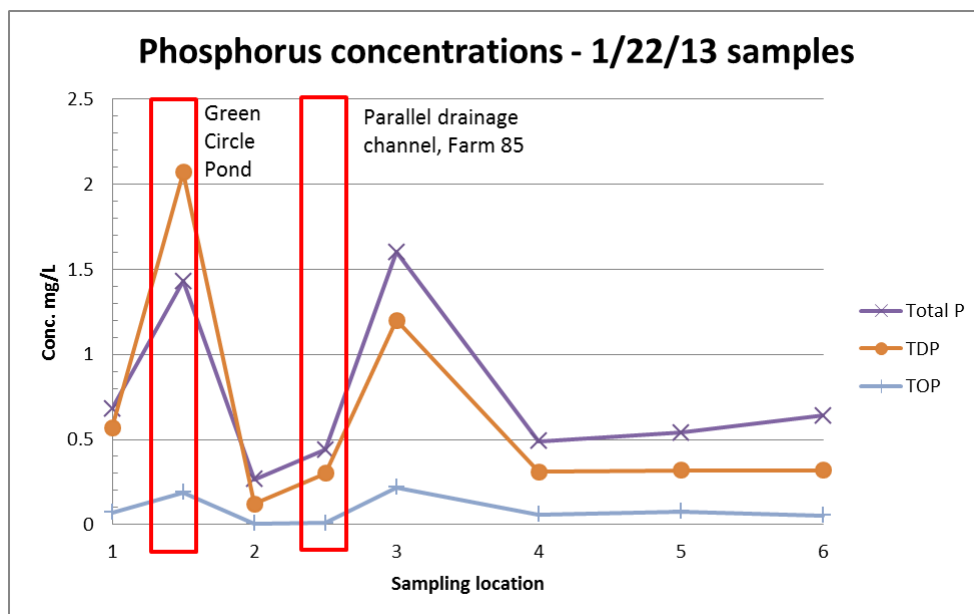


Figure 25: Phosphorus concentrations with additional points, 1/22/13.

### 3.2.2 Bacteria

Bacterial counts were observed to vary not only by location along the sampling channel, but also by date of sampling. There were also large differences in concentration between the different types of bacteria. A summary of the minimum and maximum concentrations observed on each day of sampling for each type of bacteria can be seen in Table 4.



**Table 4: Range of measured bacteria concentrations.**

<b>Bacteria type</b>	<b>1/16/13</b>		<b>1/22/13</b>	
	<b>Min (MPN/100 mL)</b>	<b>Max (MPN/100 mL)</b>	<b>Min (MPN/100 mL)</b>	<b>Max (MPN/100 mL)</b>
Total coliform	155,000	1,050,000	125,000	687,000
<i>E. coli</i>	934	45,700	1,330	166,000
Enterococci	1,790	11,600	3,650	50,400

As can be seen from the results presented in Table 4, the measured total coliform concentrations were orders of magnitude greater than those measured of either *E. coli* or enterococci. Among all types of bacteria, concentrations varied significantly along the reach of the main drainage channel. Comparisons of bacterial concentration at each sampling location on each sampling date, plotted from the most upstream to the most downstream location, can be seen in Figures 26 and 27. On both days sampled, the peak value of total coliform occurred at sampling location 3, the most downstream location on Farm 85 Trading property. The maximum concentrations of *E. coli* and enterococci occurred at location 2 on both sampling days. In general, the bacterial concentration trend along the reach of the drainage channel is similar to that of nutrient concentrations. Maximum concentrations for all observed bacterial constituents and on both sampling days occur at either location 2 or 3, with concentrations decreasing at location 4 and remaining relatively steady throughout the remainder of the channel.

Though the locations of the maxima for each bacterial constituent were temporally consistent, the magnitudes of the maximum concentrations measured on the two days of sampling were different. Under dry conditions on 1/16/13, maximum total coliform concentration was greater than 1,000,000 MPN/100 mL, while under wet conditions on 1/22/13, the maximum value was considerably lower at 687,000 MPN/100 mL. The maximum concentrations for *E. coli* and enterococci also changed between the two sampling days, but the direction of the change was opposite to that for total coliform, with greater concentrations under wet conditions than under dry, as can be seen in Table 4.

The samples collected at points outside the main drainage channel also provide information about bacterial sources in the Neo Tiew drainage area. As with the nutrient results, these additional sampling locations, referred to previously as locations 8 and 9, have been inserted into the graphs of bacterial counts along the reach of the drainage channel in order to elucidate the approximate contribution of each of the drainage areas that they represent, as can be seen in Figures 28 and 29.

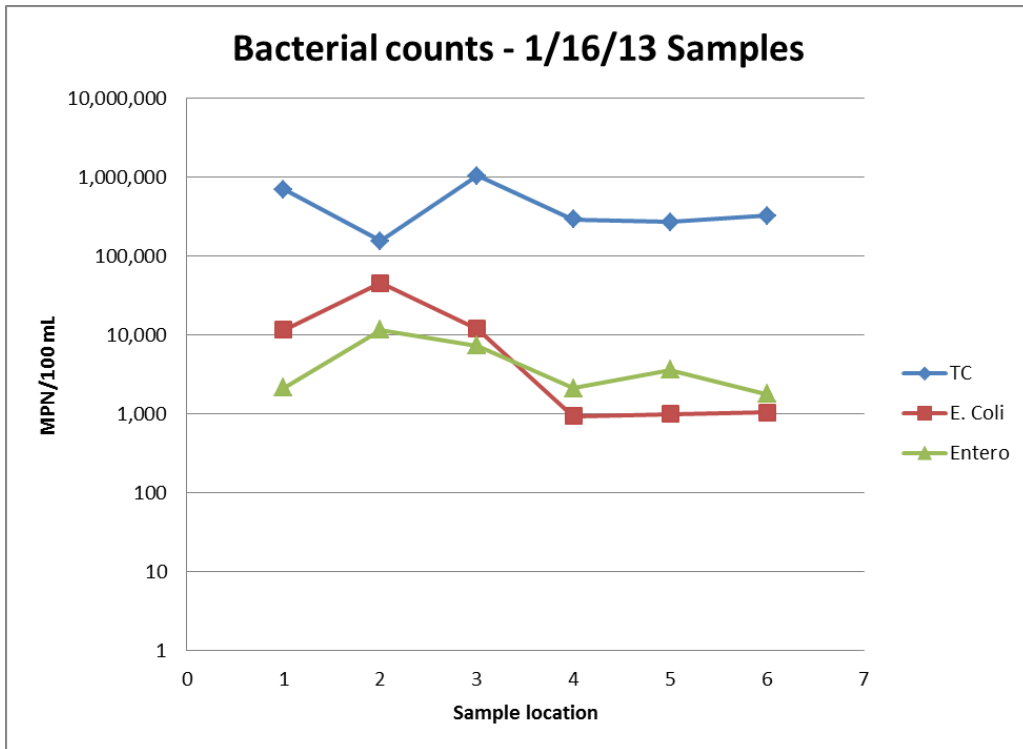


Figure 26: Bacterial counts on 1/16/13.

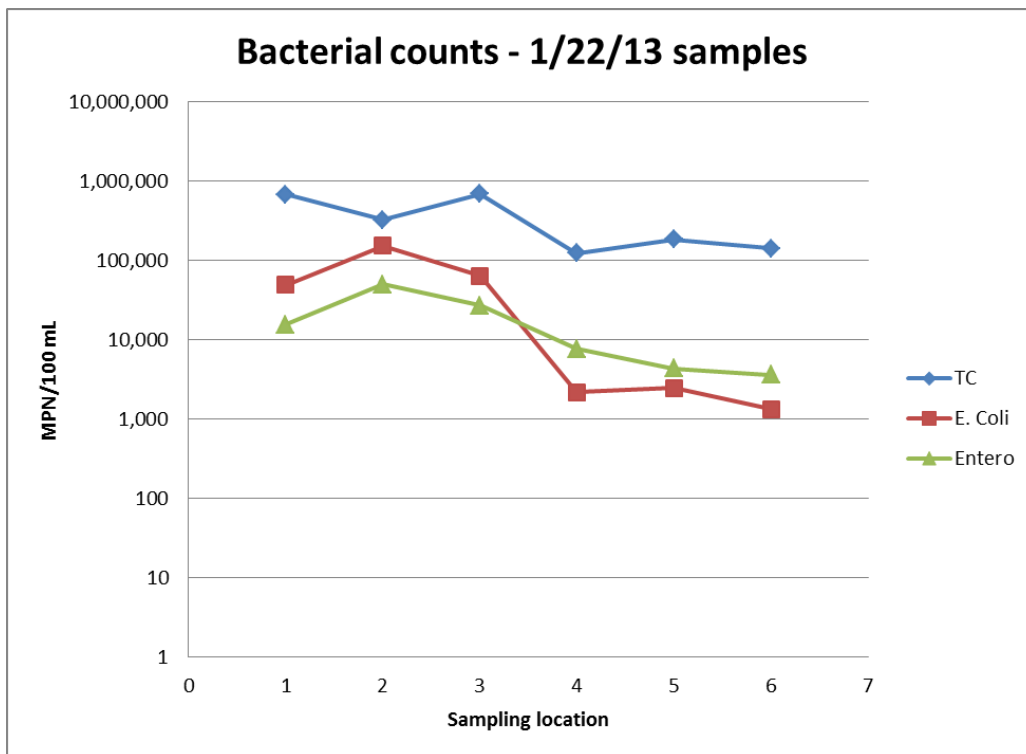


Figure 27: Bacterial counts on 1/22/13.

For all analyzed bacterial constituents, the observed concentrations in the parallel drainage channel at Farm 85 Trading were lower than those observed at the surrounding points, locations 2 and 3. Conversely, the sample taken from the pond at Green Circle Farm on 1/22/13 contained the highest observed concentrations of total coliform and enterococci, with values of 3,090,000 MPN/100 mL total coliform and 142,000 MPN/100 mL enterococci. When compared to the maximum concentrations observed within the main drainage channel, which were 1,050,000 MPN/100 mL total coliform (at location 3 on 1/16/13) and 50,400 MPN/100 mL enterococci (at location 2 on 1/22/13), it is evident that the pond at Green Circle is a particularly concentrated source of bacteria. The observed concentration of *E. coli* in the Green Circle pond was less than the value observed at location 2 on the same day of sampling.

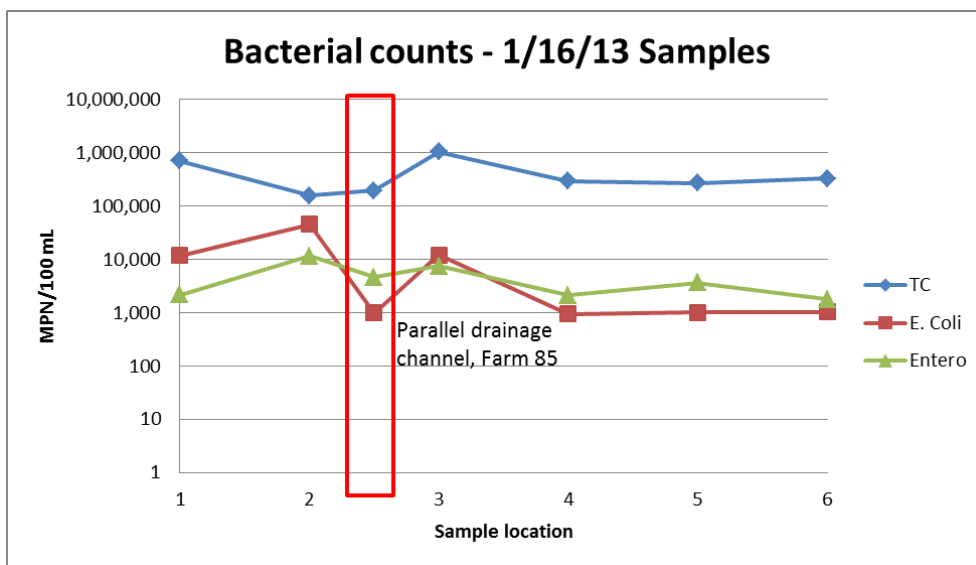


Figure 28: Bacterial counts with additional points, 1/16/13.

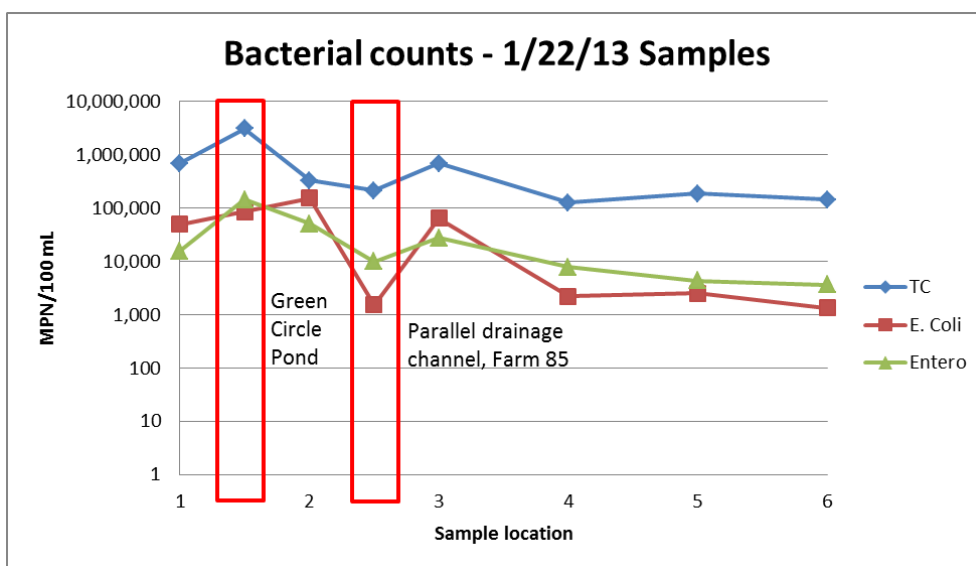


Figure 29: Bacterial counts with additional points, 1/22/13.

### 3.2.3 Solids

The concentration of total suspended solids in each water sample taken was also analyzed. The TSS concentration at each sampling location can be seen in Figures 30 and 31. Trends along the reach of the main drainage channel were consistent across both sampling days, with a strong peak solids concentration at location 3 and relatively similar and low TSS concentrations at all other sampling locations. The maximum TSS observed on 1/16/13 was 37.2 mg/L and the maximum observed on 1/22/13 was 28.4 mg/L TSS. Though more suspended solids might be anticipated under the wet conditions on 1/22/13, the maximum observed TSS concentration was greater under the dry conditions on 1/16/13.

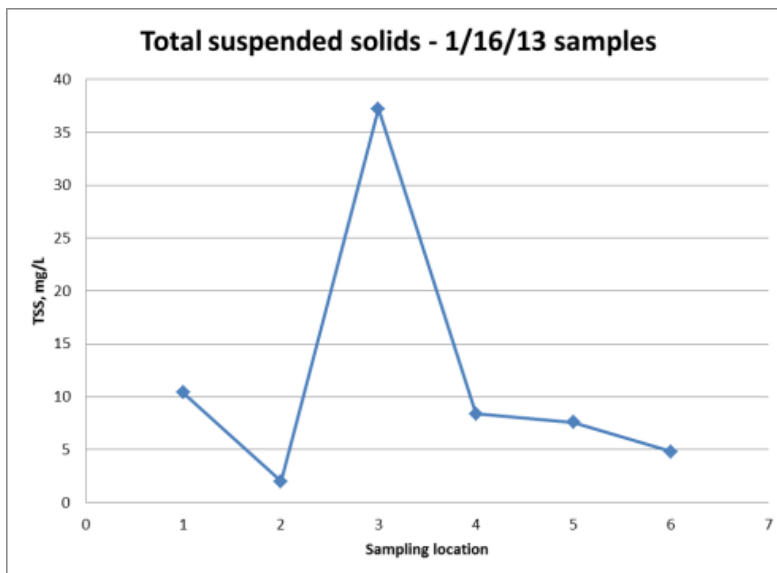


Figure 30: Total suspended solids, 1/16/13.

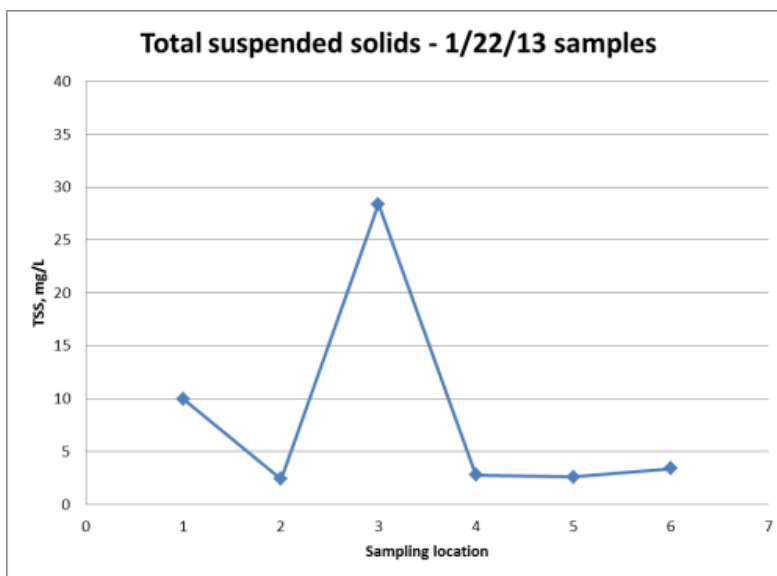


Figure 31: Total suspended solids, 1/22/13.

### 3.2.4 Field blanks and duplicates

In order to understand the reliability and repeatability of the field sampling program and the laboratory analyses procedures, field blanks were collected and analyzed and one sample was analyzed in duplicate.

Results of field blank analysis can be seen in Table 5. Concentrations for nearly all analyses on both sampling days were either below the relevant detection limit or very near the detection limit. The one notable exception, however, is the total phosphates result in the blank from 1/16/13, in which the result, 0.81 mg/L, was not only much greater than the detection limit of 0 mg/L (CHEMetrics 2012e), but it was also among the highest total phosphate concentrations observed on that day of sampling. This troubling result was not repeated in the second set of samples and analyses, however, which indicates that it was more likely the result of sample contamination or insufficiently cleaned glassware rather than a flawed sampling or analysis procedure.

Table 5: Field blank analysis results, 1/16/13 and 1/22/13.

Analyte	1/16/13	1/22/13
Total nitrogen (mg/L)	<0.01	<0.01
Nitrate-N (mg/L)	0.08	0.03
Nitrite-N (mg/L)	0.01	0.01
Ammonia-N (mg/L)	0.03	<0.01
Total phosphates (mg/L)	0.81	0.08
Total dissolved P (mg/L)	0.0081	0.015
Total organic P (mg/L)	0.09	0.099
Total suspended solids (mg/L)	<2	<2
Total coliform (MPN/100 mL)	<1	<1
<i>E. coli</i> (MPN/100 mL)	<1	<1
Enterococci (MPN/100 mL)	<1	<1

Duplicate analytical tests were performed on the sample collected at location 2 on 1/22/13 for all analyses that were performed in-house. The duplicate results can be seen in Table 6. For most nutrient analyses, the measured concentrations in the original sample and the duplicate sample were similar, with the exception of the ammonia-N result. There was a 220% discrepancy between the ammonia-N concentrations of the duplicate samples, which may be attributable to inadequate mixing of the sample prior to decanting into analytical glassware. The second sample analyzed, 2-2D, contained the volume from the bottom of the collection glassware, which would contain any settled material and thus potentially account for the significantly higher concentration of ammonia-N. The bacterial counts in the sample and the duplicate were relatively similar, with the largest variation being 15% in the *E. coli* count. In general, the results of the duplicate analyses demonstrate repeatability in the laboratory analytical procedures.

**Table 6: Duplicate sample analysis results.**

<b>Analyte</b>	<b>2-2</b>	<b>2-2D</b>	<b>% change</b>
Nitrate-N (mg/L)	1.08	1.05	3
Nitrite-N (mg/L)	0.15	0.15	0
Ammonia-N (mg/L)	0.83	2.66	-220
Total phosphates (mg/L)	0.24	0.29	-21
Total coliform (MPN/100 mL)	325,500	325,500	0
<i>E. coli</i> (MPN/100 mL)	166,400	142,100	15
Enterococci (MPN/100 mL)	49,600	51,200	-3

### 3.3 Discussion

It is evident from the results presented above that, for the majority of observed contaminants, the most concentrated inputs of nutrients and bacteria are occurring between sampling locations 2 and 3. The additional runoff entering the drainage channel between locations 2 and 3 is predominantly, if not exclusively, from the Farm 85 Trading lot. Farm 85 Trading employs intensive row-cropping practices to produce vegetables, including fertilizer application and spray irrigation. These two practices can combine to produce high volumes of runoff with high concentrations of nitrogen, phosphorus, and bacteria.

Though for most of the studied contaminants the highest concentrations were observed between locations 2 and 3, not all contaminants exhibited this behavior. Rather, ammonia nitrogen, *E. coli*, and enterococci exhibited peak concentrations between locations 1 and 2. An examination of the concentrations observed in the pond at Green Circle Farm can help to explain the high concentrations of ammonia nitrogen and fecal bacteria observed at location 2. The pond at Green Circle is used as a holding area for collected rainwater and crop runoff prior to recycling for irrigation within Green Circle Farm. Green Circle does not apply any fertilizers, synthetic or manure-based, to its crops, which explains why the concentrations of other nitrogen species and phosphorus are not elevated in the pond sample. However, there are human settlements in the immediate vicinity of the pond, which could lead to an elevated presence of ammonia nitrogen, *E. coli*, and enterococci, all of which are present in human excreta. Though observed concentrations of these contaminants were high within the pond, Green Circle captures and recycles a large portion of its crop runoff, and thus the volume of runoff from Green Circle to the main drainage channel is likely low, meaning that the Green Circle runoff is readily diluted by the flow in the main drainage channel.

Nitrification, or the successive bacterial conversion of nitrogen in the form of ammonia to nitrite and then to nitrate, has been observed to occur in channels that receive nitrogenous pollutants (Bansal 1976), much as the Neo Tiew drainage channel is receiving nitrogenous runoff. In a channel in which nitrification is occurring, one would expect to see a simultaneous decrease in concentration of ammonia-N, a temporary increase in concentration of nitrite-N, and an overall increase in concentration of nitrate-N along the reach of the channel (Bansal 1976). An examination of nitrogen species concentrations along the Neo Tiew drainage channel, shown in Figures 18 and 19, does not reveal the concentration trends associated with nitrification. This deviation from the expected trend may be due to the successive inputs of additional nitrogen contamination, of all nitrogen constituent species, along the reach of the channel. Additional

inputs of nitrogen could easily obscure the effect of nitrification. Furthermore, the hydraulic residence time from the start of the channel (location 1) to sampling location 3 is on the order of 2-3 minutes, which is not sufficient residence time to allow the nitrification process to complete. It is unlikely that nitrification is a leading cause of the observed nitrogen species concentration trends in the drainage channel.

It is useful to compare the observed contaminant concentrations to those observed by Le (2013). Le (2013) collected composite samples from 10 storm events in 2011 at a sampling location on Farm 85 Trading property within the main drainage channel, roughly at the midpoint between sampling locations 2 and 3. A comparison of the observed nutrient and solids concentrations from this study to the average event mean concentrations (EMCs) found by Le (2013) are presented in Figure 32.

The observed total nitrogen and nitrate-nitrogen concentrations at sampling location 3, which represents roughly the same drainage area as the samples collected by Le (2013), were higher under both wet and dry conditions than those observed by Le (2013), as shown in Figure 32. Conversely, the concentrations of total phosphates observed by Le (2013) were significantly higher than those observed at sampling location 3 under both dry and wet conditions. However, the observed values of total dissolved phosphorus on both sampling days are very similar to the average seen by Le (2013).

This difference in phosphorus species can be explained when considered in tandem with the much higher concentration of suspended solids observed by Le (2013), as depicted in Figure 32. As a large portion of phosphorus present in runoff is bound to solids (He et al. 2006), the water collected in this study, which has less suspended solids, is likely to have a lower total phosphate concentration than that of Le (2013) while maintaining a relatively constant concentration of dissolved phosphorus, which is independent of solids concentration. Though the samples collected in this study on 1/22/13 were under wet conditions, it had been raining for several hours prior to sample collection, meaning that the first flush of solids through the drainage channel had already occurred. Le's (2013) wet weather sampling was performed using an automatic sampler, which enabled their study to include the full duration of a given storm event, including the solids-laden first flush. This may account for the observed differences in total phosphates and total suspended solids.

### **3.3.1 Conclusions**

The results of the field collection and laboratory analysis programs of this study strongly indicate that the primary source of nitrogen, phosphorus, and total coliform contamination in the Neo Tiew drainage area is Farm 85 Trading, which resides between sampling locations 2 and 3. Thus, the most appropriate location for any remediation structure is immediately downstream of sampling location 3, as concentrations would be the highest and flow volumes are lower than at subsequent points along the channel reach. Additionally, source control measures should be directed towards the activities conducted on Farm 85 Trading property.



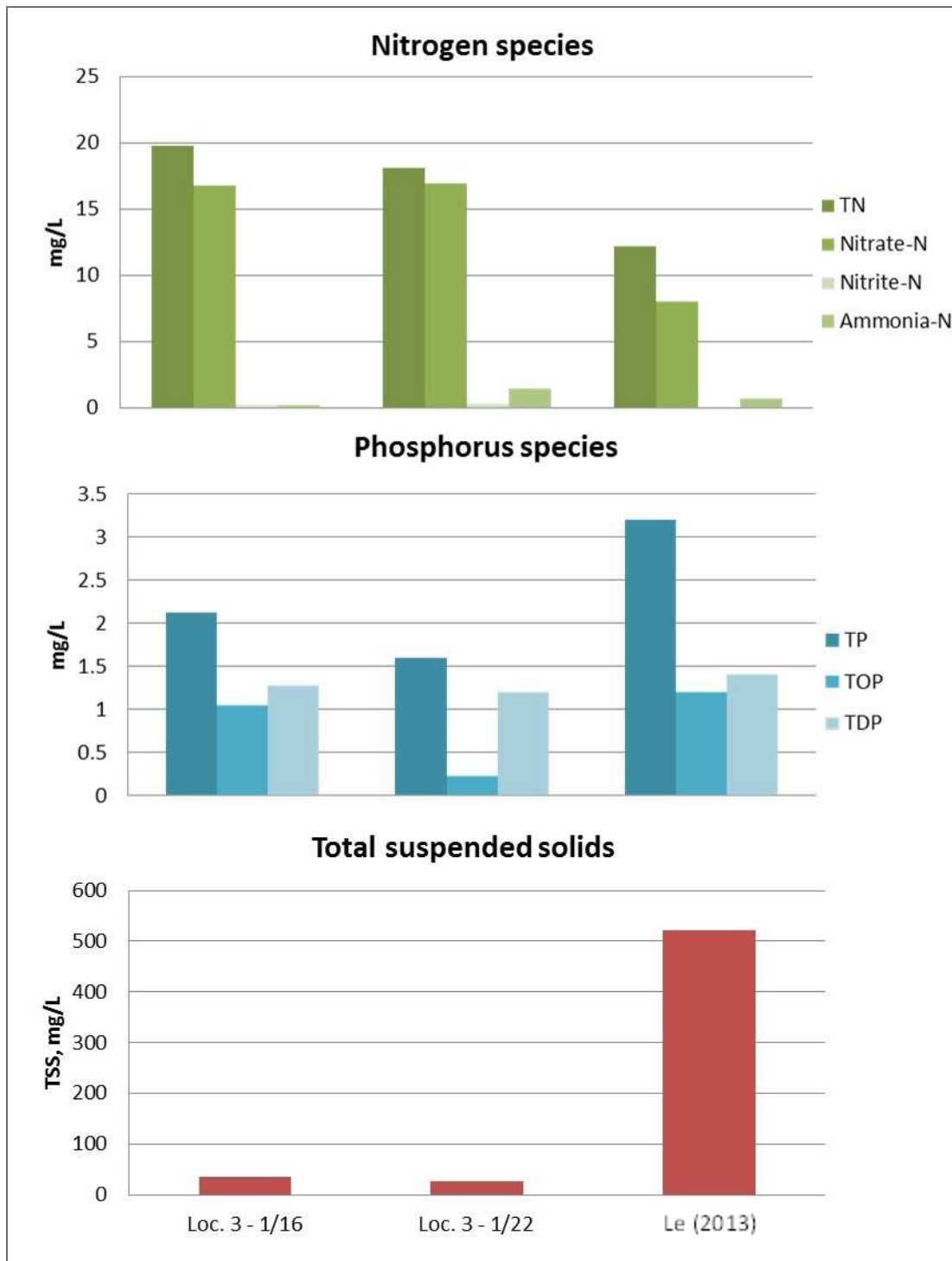


Figure 32: Comparison of observed contaminant concentrations to Le (2013).

The agricultural activities upstream of Farm 85 Trading, including two plant nurseries and one organic farm, undoubtedly contribute nutrient and bacterial contamination as well, as can be seen in the high concentrations of ammonia nitrogen, *E. coli*, and enterococci at sampling location 2. Contamination contributed by these sources would be controlled by any structural BMP installed

downstream of sampling location 3, but source control measures for these locations may need to be evaluated separately from those intended for Farm 85 Trading.

Furthermore, the comparison of the observed solids and total phosphate concentrations to those observed by Le (2013) indicates that there is a relationship between TSS and TP. It is likely possible, then, to achieve significant reductions in phosphorus concentration by reducing the concentration of suspended solids in the drainage channel.

### **3.3.2 Sources of error**

Error may have been introduced into the above results by inaccuracies in the field sampling or analysis programs. As is evident from the blank results presented above in Table 5, there was generally good process control, as reflected by observed concentrations in the blank samples near or below the relevant detection limits. However, the high concentration reading of total phosphates in the blank sample indicates that there may have been contamination of laboratory glassware or of the blank sample that compromised the accuracy of the total phosphates analysis. The glassware used for TP analysis was not washed using detergents, which removes a possible phosphate contamination pathway from consideration. Furthermore, the analytical glassware was rinsed with tap water between each sample, which, in Singapore, is unlikely to be a source of phosphate contamination (PUB 2010). Having ruled out these two potential contamination pathways, it appears that it is more likely that there was either residual phosphate material in the glassware from a previous sample or that the blank sample was contaminated in collection.

Similarly, discrepancies in the duplicate sample results (Table 6) indicate that there may have been inadequate mixing of samples prior to decanting for analysis, a procedural flaw that could have a considerable effect on contaminants which tend to be sorbed to settleable solids. Additionally, by the very nature of field collection, particularly using a grab sampling method, there is no guarantee that samples collected are representative of site conditions as a whole.

Design considerations for a best management practice will be based upon the observed conditions but, given the potential for error in analysis and variability in site conditions, must also incorporate an appropriate and reasonable factor of safety.

## 4. Best Management Practices

### 4.1 Non-structural Best Management Practices

Irrigation and fertilizer management are critical first steps in mitigating the contaminant contributions of agriculture, with structural best management practices (BMPs) to be used when source mitigation is not sufficient to achieve pollution control goals. A number of case studies demonstrate that, while potentially costly to install, irrigation management and runoff treatment systems can ultimately result in cost savings for nurseries (Alexander 1993), a finding which may be readily applicable to other types of intensive agriculture as well. Fertilizer management, irrigation management, and plant configuration have been shown by a number of studies to be effective at mitigating contaminant loads in nursery runoff. Fertilizer application can be minimized by several methods, including matching fertilizer application to plant growth cycles (Alexander 1993), fertilizer injection rather than spray application, integrated pest management to identify infestation risks (EPA 1992), and use of controlled-release fertilizers (Yeager et al. 1993).

Similarly, careful management of irrigation has been shown to minimize runoff flows and nutrient leaching from nurseries. Irrigation cycling (Bilderback 2002), drip irrigation, and irrigation scheduling (EPA 1992) have all produced positive plant-growth results and could significantly reduce any agricultural operation's water costs. Lastly, the physical arrangement of plants or crops within an agricultural area can impact the quality of runoff. By locating the most water- and fertilizer-intensive species furthest from drainage channels, nutrient loads to nearby water bodies are decreased (EPA 1992, Alexander 1993). Though these BMPs are all either low-cost or cost-saving, cultural barriers to implementation may exist and must be considered when making operational recommendations.

Organic agriculture is one prominent management practice that eliminates many types of harmful inputs, particularly synthetic fertilizers and pesticides (Brown 1993). The cultivation method relies, rather, on whole biological systems, including animal manures, plant-based compost, and biological pest control to address the agricultural problems normally solved with chemical application (Brown 1993). Though the concentrations of nutrients found in runoff from agricultural operations employing biologically-based fertilizers tend to be much lower than those observed in runoff from so-called "conventional" farms, there is frequently still a presence of excess nutrients. For example, typical nitrate losses due to leaching from a hectare of arable land under conventional cultivation range from 15-65 kg/year, while a typical value for a hectare of land under organic cultivation is 11 kg/year (Brown 1993). Furthermore, while organic farming can alleviate nutrient and pesticide contamination of storm- and surface water, it may also introduce microbial contamination through the use of animal wastes. Thus, while organic farming offers an opportunity to reduce some contaminant inputs, it must be employed in tandem with other BMPs in order to protect runoff water quality.

Yeager et al. (1993) compared nitrate-N concentrations in irrigation leachate from nurseries using controlled release fertilizers (CRF) to nurseries using CRF augmented with traditional solution fertilizers. Though nitrate-N loading rates were lower in leachate from areas using only CRF, the nitrate-N values still exceeded 10 mg/L, the drinking water standard for the chemical. Additionally, growth was significantly reduced in plants fed only by CRF (Yeager et al. 1993). CRF is much more expensive than traditional fertilizers (Bilderback 2002), so, given the

unimpressive nitrate-N reduction results and the reductions in plant growth, CRF is not likely to be a viable option for reducing nutrient loading in runoff.

Adjustments to irrigation scheduling and efficiency can improve not only the quantity of runoff from intensive agricultural operations, but also runoff water quality (Bilderback 2002). One particularly promising, low-cost system for improving irrigation efficiency is cyclic irrigation. Cyclic irrigation uses existing irrigation equipment, but rather than applying water to plants once per day in a long, continuous release, water is applied a number of times during the day for shorter times. This improves water absorption in plant substrates and thus reduces leaching of nutrients through the soil. Implementation of cyclic irrigation has been seen to reduce total irrigation volume by 44%, which represents a cost savings to operators while only reducing plant growth by 10%. Furthermore, nitrate-N concentrations in runoff were reduced by 66% and total phosphorus concentrations were reduced by 57% (Bilderback 2002). Cyclic irrigation requires no investment, as it utilizes existing equipment, and not only reduces total water inputs, but also improves runoff water quality.

#### **4.2 Structural Best Management Practices**

Several options exist for structural BMPs to remediate nursery runoff prior to discharge to collection channels. Vegetative filter strips (VFS), areas of vegetation downslope from agricultural operations, have been shown to remove over 75% of nitrogen, phosphorus, and fecal coliform from agricultural runoff (Douglas-Mankin and Okoren 2011, Abu-Zreig et al. 2003, Schmitt et al. 1999). Both surface and sub-surface flow constructed wetlands have been studied in depth for nutrient removal efficiencies (Kadlec and Knight 1996, Taylor et al. 2006, Picard et al. 2005, Huett et al. 2005, Jordan et al. 2003, Sakadevan and Bavor 1999, Jing et al. 2002, Khatiwada and Polprasert 1999, White et al. 2011). Vegetated and unvegetated sub-surface wetlands have been analyzed (Huett et al. 2005) and the effectiveness of various plant species has been studied in depth as well (Picard et al. 2005, Schmitt et al. 1999, Jing et al. 2002). Another option for remediation of irrigation leachate is a small-scale bioreactor (Wilson and Albano 2009). Each treatment system has its own limitations, including space requirements, cost, and contaminant treatment capabilities. An evaluation of site conditions will facilitate selection of the appropriate structural BMP for the Neo Tiew drainage channel.

Douglas-Mankin and Okoren (2011) studied the ability of a 150 m-long VFS to remove nutrients and bacteria from feedlot runoff. Remediation by VFS was shown to be effective in removing solids, nutrients, and fecal coliform. Total nitrogen concentration was reduced by 77% and phosphorus was reduced by 84%. Additionally, the vegetated buffer significantly reduced bacterial contamination, with removal rates of 83.5% for fecal coliform and 91% for *E. coli* (Douglas-Mankin and Okoren 2011). Though the study was of remediation of feedlot runoff rather than nursery or vegetable crop leachate, the impressive reduction results can be readily applied to other agricultural contexts.

There are a number of options for dimensions, slope, and vegetation type for VFS implementation. Abu-Zreig et al. (2003) varied length, slope, and vegetative cover in their study of phosphorus removal in 20 field-scale VFSs. Phosphorus was removed in all of the various configurations studied, with total P removal ranging between 31% and 89%. Length was determined to be the primary determining factor in phosphorus removal efficiency, with the 89% removal rate in a 15-m-long filter. Shorter filters were effective at removing sediment and

sediment-sorbed contaminants, but longer filters were necessary to remove dissolved contaminants through infiltration and plant adsorption (Abu-Zreig et al. 2003).

In a similar study of field-scale VFS configurations, Schmitt et al. (1999) analyzed the remediation performance of VFSs by varying vegetation type, width, and contaminant profile. The phosphorus reductions observed in their study (55% to 79% of total phosphorus removed) agree with those determined by Abu-Zreig et al. (2003) and they also found nitrate-N removal capacities ranging between 24% and 48%. Schmitt et al. (1999) likewise found width to be an important factor in contaminant removal, with infiltration in a 15 m-wide filter twice as high as that in a 7.5 m-wide filter, though sedimentation was not improved with the additional width. Grass was determined to be the optimal vegetation for removing sediment-sorbed contaminants and the inclusion of trees and shrubs in filter vegetation was not found to improve performance (Schmitt et al. 1999).

Constructed wetlands are an effective form of remediation for a variety of wastewater sources and can be particularly useful in the reduction of contaminant loads in agricultural runoff (Kadlec and Knight 1996). Constructed wetland systems are generally divided into two categories: free-water-surface (FWS) and subsurface-flow systems (SFS). In FWS wetlands, shallow depths of water flow through vegetation and in SFS wetlands, water flows through a sand or gravel substrate below the ground surface (EPA 1988).

In a field study of a large, established FWS wetland, Taylor et al. (2006) observed high rates of nitrate-N removal. The wetland studied had an areal extent of 3.8 hectares vegetated with a wide diversity of plant species and sustained a typical flow of 2.1 million liters per day. The removal rates varied seasonally, with average removal from spring through fall of 94.7% and 70.7% in the winter season. There was no observable trend for phosphate-P reduction, with phosphate-P concentrations sometimes actually increasing after flow through the wetland (Taylor et al. 2006).

Jordan et al.'s (2003) study of a 1.3-ha restored wetland found significant potential for nutrient removal, with 59% of total phosphorus and 38% of total nitrogen removed during the first year of observation. However, during the second year of the study, there was a great deal of fluctuation in flow and influent contaminant concentrations, the combination of which resulted in low removal rates. In general, removal rates were observed to increase with greater influent contaminant concentrations, an observation which is supported by the widely-held theory that nutrient removal can be approximated by first-order kinetics (Jordan et al. 2003). Overall, the removal rates observed in the Jordan et al. (2003) study were lower than the average removal capabilities reported by Kadlec and Knight (1996), which may indicate that wetlands operate best under conditions of relatively consistent flow rates and high contaminant concentrations.

Several other studies have observed wide ranges in the nutrient removal capabilities of wetlands under different conditions, including those by Sakadevan and Bavor (1999) and Jing et al. (2002). Sakadevan and Bavor (1999) varied influent contaminant concentrations and hydraulic loading rates in laboratory-scale constructed wetlands and observed nitrogen and phosphorus removal rates. Nitrogen removal ranged from 26.3% to 77.5% and phosphorus removal ranged from 11% to 48.9% under variable conditions. The results indicate that lower hydraulic loading rates led to greater removal of nutrients, though other factors contributed to removal efficiency as well (Sakadevan and Bavor 1999). Jing et al. (2002) constructed microcosm wetlands and varied hydraulic residence time and plant species. Similarly to Sakadevan and Bavor (1999), Jing

et al. (2002) concluded that hydraulic loading rate (and therefore hydraulic residence time) greatly affected nutrient removal, with peak removal rates of 80% of ammonia nitrogen and 46% of organic phosphorus at a hydraulic residence time of 3 days.

Huett et al. (2005) simulated sub-tropical nursery runoff flow through constructed vegetated and unvegetated subsurface treatment wetlands in order to observe nutrient removal rates. Both vegetated and unvegetated systems were built with a gravel substrate and the vegetated cells were planted with *Phragmites australis*, a species of reed. The study demonstrated the importance of vegetation in nutrient removal, with vegetated wetland nutrient attenuation rates of greater than 96% for both phosphorus and nitrogen. By contrast, the unvegetated system removed less than 16% of nitrogen and less than 45% of phosphorus (Huett et al. 2005). While vegetated cells demonstrated greater nutrient removal capacity, maintenance requirements are also greater than in unvegetated cells. Vegetation must be harvested periodically to prevent its decomposition in the wetland cell, which would release the captured nutrients back into the water.

While Huett et al. (2005) demonstrated the importance of vegetation in water treatment wetlands, Picard et al. (2005) observed constructed FWS wetland systems in laboratory conditions in order to understand the influence of vegetation type and temperature on nutrient removal rates. Removal rates varied with plant species and temperature, with removal as high as 90% for nitrogen and 74% for phosphorus, but “no clearly defined pattern of species-specific nutrient removal was found” (Picard et al. 2005). This finding is confirmed by a number of other studies, including Jing et al. (2002) and Kadlec and Knight (1996), both of which found that, while wetland vegetation is important for removal of nutrients and bacteria from runoff, the type or species of vegetation does not affect wetland treatment effectiveness. In contrast, Tyler et al. (2012) found that while all vegetation species under observation in their study decreased dissolved phosphate concentrations by more than 90%, mesocosms planted with grass or cattail achieved the reduction more quickly than those planted with reed. In general, it has been shown that surface flow and subsurface flow wetlands are most effective at contaminant removal when planted with macrophytes that are stable in the local environment and wetland ecosystem, regardless of species (Kadlec and Knight 1996).

Given the importance of phosphorus to the Kranji ecosystem, the study of phosphorus reductions from nursery runoff in SFS wetlands by White et al. (2011) is highly relevant. The study, which included laboratory and field observations, compared phosphorus reductions across three different clay substrates. Phosphorus reductions of up to 76% were observed using coarse calcined clay, which was determined to be the most effective substrate. Across all three substrates (crushed red brick, coarse calcined clay, and fine calcined clay), reductions of between 60% and 74% were observed for up to seven months, at which point the system became saturated with phosphorus. After saturation, slow phosphorus desorption could restore the system’s remediation capacity, but this requires a period of rest for the wetland system (White et al. 2011).

While others studied the short-term effectiveness of nutrient removal in constructed wetlands, Borin and Tocchetto (2007) observed the removal of nitrogen in a constructed wetland over a five-year period. They observed a 0.32-ha surface-flow wetland that drained an area in which a number of different commodity crops were cultivated. Over the total study period, the wetland removed approximately 90% of nitrogen from the agricultural runoff, though loading and removal rates varied considerably on shorter time scales (Borin and Tocchetto 2007). Though

nutrient removal efficiency may fluctuate according to different variables, it is evident that constructed wetlands are generally highly effective at alleviating nutrient contamination in agricultural wastewater.

Constructed wetlands are not only effective at removing nutrients and solids from wastewater, but have been demonstrated to be capable of achieving significant reductions of fecal coliform as well. Khatiwada and Polprasert (1999) report that FWS constructed wetlands remove microorganisms from tropical runoff by solar radiation, sedimentation, adsorption, filtration, and increased water temperature. This removal is modeled by first order kinetics, so removal rates depend upon influent concentration and hydraulic residence time within the wetland, but it is apparent that significant reductions in bacterial concentrations by constructed wetlands are possible (Khatiwada and Polprasert 1999).

In contrast to the natural treatment systems presented above, the constructed wetland system that Wilson and Albano (2009) studied is essentially a small-scale manmade bioreactor that, while demonstrated to be effective at removing nitrogen from nursery runoff, requires much greater initial investment and continuous maintenance. The system is a series of small tanks filled with Kaldnes media, a proprietary plastic filter media, seeded with attached-growth microorganisms. The studied system demonstrated nitrate removal capacities of greater than 90%, but can only accept a flow of approximately 10 L per minute. This bioreactor system would be effective for land-limited areas but has high maintenance requirements and costs (Wilson and Albano 2009).

Clearly, there are a number of potential methods to reduce nutrient and bacterial loading from nursery runoff. By characterizing nursery conditions, including quantity and quality of leachate, as well as limitations for the design of structural BMPs, including physical space, land ownership, and natural vegetation, the large pool of remediation systems can be restricted to a few applicable options. Furthermore, cost, maintenance, and cultural considerations must be taken into account. Evaluation of these factors was included as part of the field work conducted for this study.

#### **4.3 BMP selection**

Though there are benefits and detriments associated with each type of structural BMP, some are more appropriately suited to the conditions observed in the Neo Tiew subcatchment. In particular, the wide range of treatment capabilities of constructed wetlands matches the wide range of contaminant profiles observed in the area and the large fluctuation in flow between dry and wet conditions. Constructed wetlands are effective at reducing many types of contaminants over a range of concentrations through a combination of physical, chemical, and biological processes (Sharma et al. 2008). A FWS wetland is more appropriate for the Neo Tiew subcatchment than a SFS wetland as free-water-surface wetlands require less excavation to construct than subsurface-flow wetlands and are generally easier to operate and maintain (Kadlec and Knight 1996). Given the selection of a free-water-surface constructed wetland as the most appropriate structural BMP for the Neo Tiew subcatchment, the wetland must be designed in order to meet treatment goals, comply with local regulations, and conform to site conditions.

#### **4.4 Structural BMP design**

There are many possible configurations of constructed wetlands, each appropriate to a unique set of treatment requirements and site conditions. Though an ideal treatment wetland would completely remove all contaminants from influent stormwater and fit seamlessly into the



landscape, in reality, design choices must be made to find the most suitable compromise among performance, cost, and acceptability. Primary design considerations for any constructed wetland include size, treatment components, vegetation, and topography.

#### **4.4.1 Wetland sizing**

A constructed wetland must be of an adequate volume and area to achieve an acceptable level of treatment for a given quantity of water. There are a number of possible methods by which to determine the design volume and treatment level, each of which prioritizes a different design objective and each of which must be considered for performance and feasibility.

##### **4.4.1.1 Sizing guidelines**

###### **4.4.1.1.1 Design storm**

Many municipalities and regulatory agencies, including the PUB, use design-storm guidelines to size drainage structures. The design-storm principle utilizes historical rainfall data and recurrence frequencies to determine the statistical return period of a precipitation event of given duration and rainfall intensity (Watt and Marsalek 2013). PUB, and other similar agencies, then set a required design return period for drainage structures in an attempt to balance sustainability, public safety, and cost. PUB specifies the design-storm return period for a given surface water drainage structure based upon the area of the subcatchment that the structure serves, as drainage structures serving larger watersheds may be more critical to public or environmental safety and thus require a longer return period for design (PUB 2011). For the Neo Tiew subcatchment, which covers approximately 28 ha, the PUB recommends a design storm with a return period of 10 years (PUB 2011).

The duration of a design storm is equal to the time of concentration of the subcatchment being served, which is the time required for a drop of precipitation to travel from the most remote part of a subcatchment to the outfall, including both overland and channel flow (PUB 2011). Given a required return frequency and design-storm duration, design-storm rainfall intensity can then be determined using an intensity-duration-frequency (IDF) curve. IDF curves are location-specific, as the statistical characteristics of rainfall vary significantly with geography and climate.

The total volume that the treatment wetland needs to contain can then be determined by using the rational method along with the design-storm duration and rainfall intensity, as established by subcatchment hydrology and the appropriate IDF curve. The rational method employs a simple formula to calculate peak runoff from a storm event by multiplying the average rainfall intensity by the catchment drainage area and a runoff coefficient, which accounts for infiltration, antecedent moisture conditions, and other non-ideal effects (Watt and Marsalek 2013). Total discharge volume from the design storm is then calculated by multiplying this peak runoff flow rate by the design storm duration, which is equal to the time of concentration for the subcatchment.

Further information is needed to determine the required wetland dimensions, however, because surface area and depth are critical factors in wetland design (Kadlec and Knight 1996). One possible method to translate design storm volume into wetland area is to use a typical wetland depth and aspect ratio in order to establish the external dimensions of the wetland. Kadlec and Knight (1996) use an average wetland depth of 0.45 m to size a wetland using a design storm. They also recommend a minimum length-to-width ratio of 2 to 1, which can be used in

conjunction with the typical depth value to determine the wetland's external dimensions. This minimum aspect ratio is intended to encourage flow and effluent distribution, which will be further improved with a greater aspect ratio (Kadlec and Knight 1996). Once this initial sizing has been set, the actual volume of water contained by the wetland will likely increase as wetland components are designed, as many wetland features incorporate regions of deep flow, while design depth throughout shallow areas of the wetland is rarely less than the typical 0.45-m design depth (Kadlec and Knight 1996).

#### 4.4.1.1.2 Scale to reference area

Another method for determining the required area of a treatment wetland is to scale the wetland area according to a given reference wetland and watershed area. This method is simple and can be beneficial in wetlands and watersheds that closely resemble existing wetlands that have been demonstrated to achieve a required level of treatment. A number of constructed wetland guidelines and studies employ this method, including Dupoldt et al. (1993) and Millhollon et al. (2009), in which the authors sized treatment wetlands by scaling the area proportionally in comparison to reference watersheds and wetlands. The proportions recommended by Dupoldt et al. (1993) result in constructed wetlands ranging in size from 1.8 to 2.6% of total wetland area.

Though this method is simple and facilitates comparison across different watersheds and wetlands, it may present drawbacks as a sole design criterion for wetland sizing. First, in very large watersheds, the area recommended by this linear scaling may become prohibitively large for implementation. Furthermore, the simplistic sizing calculation does not account for variations in hydrology or climatic characteristics in diverse locations and thus may result in either over- or under-sizing a wetland, depending on site characteristics. Lastly, there is evidence across many wetland studies that indicate there exists a wide variety of wetland-to-watershed area ratios that achieve adequate treatment, making selection of an appropriate scaling factor difficult. While comparison to a reference ratio or a percentage of total watershed area may not be a sufficient design method on its own, it can be a helpful tool to verify the results of other design methods.

#### 4.4.1.1.3 Residence time

The ability of a constructed wetland to reduce the concentration of contaminants in surface water runoff to a specified level is, needless to say, a critical aspect of the wetland's performance. As nutrient and bacteria removal in constructed wetlands is roughly represented by first-order reaction kinetics (Kadlec and Knight 1996, Khatiwada and Polprasert 1999), in which the removal of contaminants is directly dependent upon the time that a parcel of water spends in the treatment wetland, it is evident that the hydraulic residence time of a wetland is an indicator of effluent water quality (Kadlec and Knight 1996).

Given the non-ideality of natural systems, determining the hydraulic residence time of a constructed wetland is more complicated than simply knowing the dimensions and the flow through the system. Inaccuracies can arise from irregularities in wetland depth and volume, as well as non-ideal mixing and detention. One method for calculating a more accurate hydraulic residence time is using a residence time distribution (RTD), which essentially applies a probability density function to the hydraulic residence time of a parcel of water in a constructed wetland. However, the true RTD for a given wetland can only be determined by using a tracer study after the wetland has been constructed (Kadlec and Knight 1996). Thus, using hydraulic residence time as a front-end design parameter in sizing a wetland is limited to, at best, an approximation of non-ideal wetland conditions by incorporating values from similar constructed

wetlands (Kadlec and Knight 1996). Hydraulic residence time is an important design consideration, but care must be taken to account for the imperfections of this calculation in constructed wetland systems.

Accepted values of first-order removal rates for given contaminants in constructed treatment wetlands are available in a number of studies, including fecal coliform removal as studied by Khatiwada and Polprasert (1999) and phosphorus, nitrogen, and microorganism removal as presented by Kadlec and Knight (1996). These values can be used in conjunction with regulatory effluent water quality standards or design quality targets to determine the required hydraulic residence time of a given wetland. Hydraulic residence time can be translated into a total required storage volume by using a representative influent flow rate for the subcatchment being served. This design flow need not be the design storm flow rate outlined previously, but should rather be the value most representative of average conditions within the watershed and wetland. Volume can then be related to wetland area using the typical depth and aspect ratio values described in the discussion of the design storm method.

This design method incorporates wetland removal processes and performance in a way that neither the design storm nor reference ratio method does, but it presents some limitations of its own. Determining the appropriate design flow for setting a wetland volume sufficient to achieve the required residence time is difficult without an historical flow record. Sizing for wet or dry influent flow conditions can easily result in vastly different wetland volumes. Furthermore, sizing a wetland to achieve a high removal rate of a contaminant that follows slow reaction kinetics, such as fecal coliform (Khatiwada and Polprasert 1999), requires an enormous wetland volume that may be infeasible to implement. Additionally, regulatory standards for nutrients and microorganisms in surface water do not exist in many places, including Singapore, meaning that any target reduction in contaminant concentrations must be determined individually if this design method is employed.

#### **4.4.1.2 Size determination**

##### **4.4.1.2.1 Method selection**

Each of the wetland sizing methods outlined above relates wetland design to an important design consideration, including hydrology, watershed area, and removal performance. In addition to these technical design inputs, regulatory considerations must also be incorporated into the sizing of the treatment wetland. Though Singapore does not have regulatory targets for the effluent quality of a constructed wetland or for nutrient and bacterial contamination of surface water, it does require that surface water drainage structures be sized to accommodate the design storm of the appropriate return period (PUB 2011). Additionally, Singapore's tropical climate, rainfall patterns, and hydrology are significantly different than the conditions in many of the locations that have been studied in order to establish reference wetland sizes and contaminant removal rates (Millhollon et al. 2009). Given these factors, the design storm method is the most appropriate sizing method for the Neo Tiew subcatchment, but the important design considerations introduced by the other design procedures must also be kept in mind throughout the design process.

#### 4.4.1.2.2 Calculations

##### *Subcatchment Hydrology*

The PUB design-storm return period requirement for a surface water drainage structure serving a watershed area of less than 100 hectares is 10 years (PUB 2011). Though it is evident from initial approximations that the area of the Neo Tiew subcatchment is significantly less than 100 hectares, a more precise determination of watershed area is necessary for wetland sizing calculations. Le (2013) delineated the contributing area to their sampling location through a combination of field observations and topographic analysis. The watershed area contributing runoff to Le's (2013) sampling location is depicted in Figure 33.



**Figure 33: Subcatchment area from Le (2013).**

The critical location for a BMP was determined previously to be immediately downstream of sampling location 3, which is downstream of Le's (2013) sampling location and therefore receives runoff from additional area. In determining the new subcatchment area, the upstream subcatchment boundaries delineated by Le (2013) remain unchanged, but the boundary closer to sampling location 3 must be amended according to field observations and topographic data. In some locations, existing infrastructure must also be considered when delineating the watershed. For example, along the property line between Farm 85 Trading and the sand storage area (Figure 3), a solid fence forces runoff to flow in a direction that would not be predicted based on topography alone. Additionally, the network of smaller drains and channels throughout the subcatchment informs delineation of certain parts of the watershed. Combining the observations of Le (2013) and these considerations, the boundaries of the watershed that contributes to runoff at sampling location 3 were determined as depicted in Figure 34. The area of the subcatchment draining to location 3 is 28 ha, which means that the required design storm return period is 10 years according to PUB guidelines (PUB 2011).



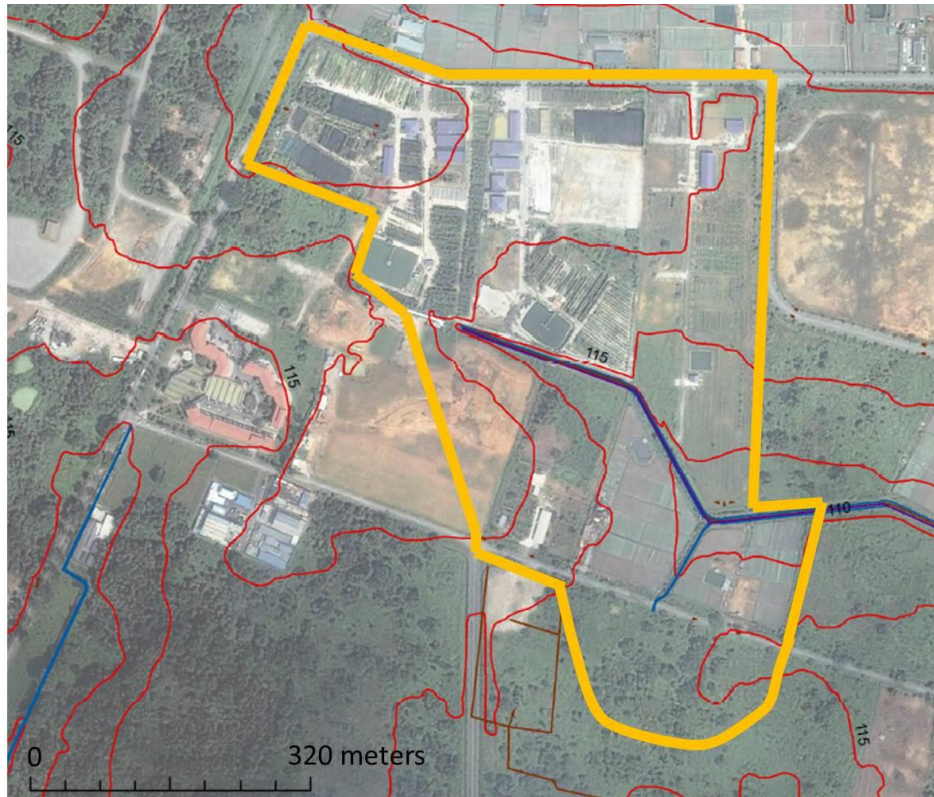


Figure 34: Subcatchment area contributing runoff to sampling location 3.

The time of concentration in the subcatchment must also be determined in order to calculate the design-storm requirements, as the duration of the design storm is equal to the time of concentration in the watershed (PUB 2011). The time of concentration was determined by evaluating the length and elevation change of the longest potential flow paths within the subcatchment, and then employing a time-of-concentration nomograph for small drainage basins (VDCR 1992). This method provides the normalized overland and channel flow times for each path within the subcatchment, and the greatest among these is the time of concentration. The longest flow time within the Neo Tiew subcatchment, and thus the time of concentration for the watershed, was determined to be 10 minutes.

#### *Design storm*

Intensity-duration-frequency (IDF) curves are used to determine the characteristics of a design storm. Given two of the characteristics (intensity, duration, or frequency), the third can be readily determined. Using the IDF curve for Singapore (PUB 2011), shown in Figure 35, with a return period of 10 years and a duration of 10 minutes, the intensity of the required design storm was determined to be 195 mm/hr.

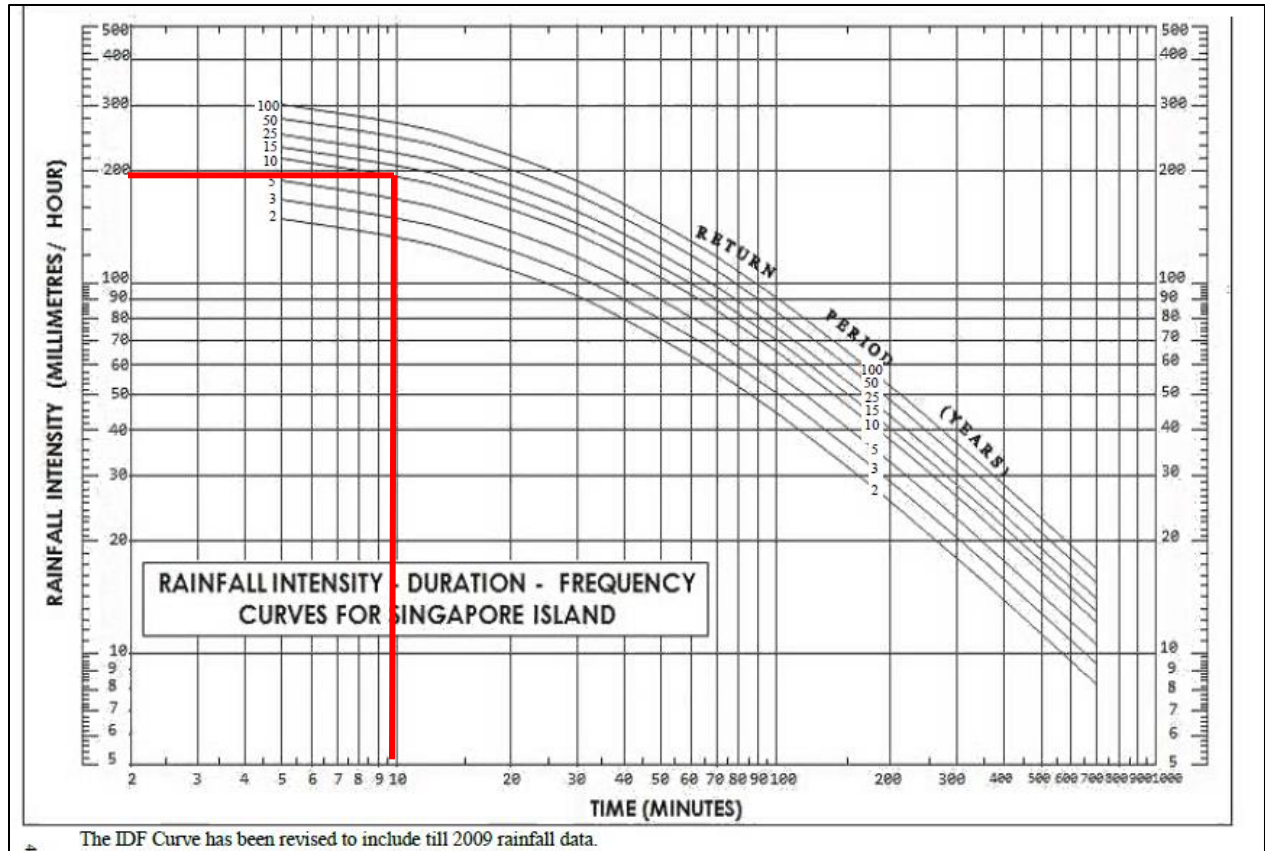


Figure 35: Singapore IDF curve (PUB 2011).

Intensity and watershed area are translated into a peak design storm runoff flow rate using the rational method, which relates the parameters as shown in Equation 3. (PUB 2011):

$$Q_r = \frac{1}{360} ciA \quad (3)$$

In which:

- $Q_r$  = peak runoff at the outlet point ( $\text{m}^3/\text{s}$ )
- $c$  = runoff coefficient
- $i$  = rainfall intensity ( $\text{mm/hr}$ )
- $A$  = catchment area (hectares)

The runoff coefficient,  $c$ , is used to account for the state of development and land use within a catchment. For land under agricultural production, such as the Neo Tiew subcatchment, PUB (2011) recommends using a coefficient value of 0.45. Using this value for  $c$  and the determined values of rainfall intensity and catchment area, the peak runoff from the 10-year design storm in the Neo Tiew subcatchment is  $7 \text{ m}^3/\text{s}$ .

Given the design objective to contain the 10-year design storm, the required wetland volume can then be calculated using the peak flow rate and the storm duration. With storm duration equal to the time of concentration (10 minutes) and a peak flow rate of  $7 \text{ m}^3/\text{s}$ , the required wetland volume is  $4,200 \text{ m}^3$ .

### *Wetland dimensions*

The required design volume of 4,200 m<sup>3</sup> is used in conjunction with typical values of wetland flow depth and aspect ratio to determine the required size and shape of the wetland. Kadlec and Knight (1996) recommend a design depth in the shallow portion of the wetland of 0.45 m. This depth value is supported by other studies, including Millhollon et al. (2009), in which the shallow wetland depth was 0.53 m, and Borin and Tocchetto (2007), in which the wetland floor was 0.4 m below the surrounding field surface.

The wetland design volume, 4,200 m<sup>3</sup>, is divided by the design depth, 0.45 m, to obtain a total wetland area of 9,300 m<sup>2</sup>, or 0.93 ha. This area can further be translated into wetland dimensions by adopting the length-to-width ratio recommended by Kadlec and Knight (1996) of 2 to 1. In order to maintain the minimum required area of 9,300 m<sup>2</sup> and the minimum aspect ratio of 2 to 1, the most appropriate external dimensions for the wetland are 140 m long by 70 m wide. These dimensions result in a total area of 9,800 m<sup>2</sup>, but some of the area within those external dimensions will be occupied by berms and other non-treatment components of the wetland. With these dimensions, the wetland area is 3.5% of the total wetland area, which is well within the range of wetland to watershed area comparisons observed within the body of literature (Table 7).

With wetland dimensions determined, the mean hydraulic residence time and hydraulic loading rate for the wetland can also be calculated. These values are important points of comparison to wetlands studied in the literature, as both can be indicators of the treatment capability of the wetland (Kadlec and Knight 1996). Though determination of a true hydraulic residence time can be difficult, an annual mean residence time can be calculated by dividing the wetland volume by the total annual runoff volume. The annual runoff volume can be calculated by a method recommended by Kadlec and Knight (1996) by analogy with the rational method, in which total annual precipitation is used in lieu of rainfall intensity. The average annual precipitation in Singapore is 2,400 mm (Tan et al. 2009). Using the same runoff coefficient and catchment area as above (0.45 and 28 ha, respectively), the annual runoff volume is 302,400 m<sup>3</sup>. This gives a mean hydraulic residence time of 5 days, which is consistent with wetlands studied, as shown in Table 7. Hydraulic loading rate is calculated similarly, by dividing the total annual runoff by the wetland area (Kadlec and Knight 1996). The mean hydraulic loading rate of the wetland is 8.5 cm/day.

**Table 7: Comparison of design parameters to studied wetlands.**

<b>Study</b>	<b>Area of wetland per watershed area (%)</b>	<b>Mean hydraulic residence time (days)</b>
Borin and Tocchetto (2007)	5	≥7
Millhollon et al. (2009)	1.9	not provided
Khathiwada and Polprasert (1999)	n/a (lab-scale)	6
Jordan et al. (2003)	9	12-19
Jing et al. (2002)	n/a (lab-scale)	2-6
Sakadevan and Bavor (1999)	n/a (lab-scale)	5
Dupoldt et al. (1993)	1.8-2.6	not provided
Kadlec and Knight (1996)	8	11
This study	3.5	5



#### **4.4.2 Wetland features**

Wetlands are formed by a combination of structural components and vegetation features that together comprise a full treatment wetland. Each wetland feature serves a treatment purpose or structural function and thus must all operate together in order to achieve total treatment goals. Key features of FWS wetlands include a sedimentation forebay, a shallow vegetated region, a deep finishing pond, and a vegetated spillway.

##### **4.4.2.1 Treatment components**

###### **4.4.2.1.1 Sedimentation basin**

In the majority of treatment wetlands, a sedimentation basin, or forebay, is the first component after the inlet to the wetland. As the name indicates, the primary function of a sedimentation basin is to remove large particles and sediment from the wetland influent in order to provide the first level of treatment and also to protect downstream wetland components from these larger solids (Dupoldt et al. 1993). Sedimentation is a key removal mechanism for contaminants that are sorbed to solids, including microorganisms (Khatiwada and Polprasert 1999) and phosphorus (Kadlec and Knight 1996). Forebays are deeper than the rest of the wetland and large enough to facilitate a hydraulic residence time sufficient to achieve settling. Given these conditions, sedimentation basins also serve the secondary purpose of regulating flow in large precipitation events, buffering the rest of the wetland from large flows and flushing, both of which can be detrimental to wetland sustainability and treatment capability (Dupoldt et al. 1993).

Sedimentation basins are constructed by a combination of excavating below the surrounding field surface level and building berms above the field level. In general, the bottom floor of a sedimentation basin should be at least 3 m wide to ensure access by equipment for maintenance (Dupoldt et al. 1993). Basin floors are typically 2 m below the ground level of surrounding areas in order to achieve sufficiently still water to encourage sedimentation (Millhollon et al. 2009). The berms surrounding the basin and the sides of the excavated region of the basin should not exceed a 2 to 1 (horizontal to vertical) slope and should be vegetated in order to maintain slope stability (Dupoldt et al. 1993). The berm height can be determined in the same way as other external berms are sized, as outlined in the following discussion of berm design. Outflows from a sedimentation basin to the next wetland component should be designed to encourage distributed flow rather than a point outflow. This can be achieved by a series of multiple closely-spaced inlets or a broad-crested inlet weir, both of which encourage sheet flow through the subsequent wetland components (Kadlec and Knight 1996).

###### **4.4.2.1.2 Shallow vegetated wetland**

Another critical wetland component is the shallow, vegetated region of the wetland in which nutrients are taken up by plants (Kadlec and Knight 1996) and microorganisms are removed by solar radiation and filtration (Khatiwada and Polprasert 1999). This region is not excavated below the surrounding ground level, but is rather surrounded by external berms that both maintain the appropriate water level and protect the wetland from lateral inflows that have not passed through the sedimentation basin (Kadlec and Knight 1996). Exterior berms should be sized to “equal the sum of the maximum desired normal water level, the design storm rainfall, and the lifetime loss of freeboard due to sediment and plant accumulation, approximately 1 cm/yr” (Kadlec and Knight 1996).

The shallow flow region of the wetland should also include lower, internal berms that create long, narrow flow paths through the wetland (Borin and Tocchetto 2007). Long and narrow channels increase the aspect ratio of the wetland by functionally increasing the length of the treatment region of the wetland. Furthermore, flow paths of this nature approximate plug flow better than short, wide paths, and engender uniform treatment by preventing flow from short-circuiting the wetland (Kadlec and Knight 1996). Though there is an overall slope through the wetland, the flow channels should be level from side to side in order to further discourage preferential flow paths through the wetland (Dupoldt et al. 1993).

For optimal treatment performance, the long and narrow flow channels created by internal berms in the shallow wetland region should be intersected at various points by unvegetated areas of deeper water arranged perpendicular to the path of flow. These deep transverse ditches improve flow distribution and residence time, both of which have been shown to improve nitrogen removal in constructed wetlands (Kadlec and Knight 1996). In order to achieve sufficient depth for flow distribution, these ditches should be excavated 1 to 2 m below the floor of the shallow wetland.

#### 4.4.2.1.3 Deep finishing pond

Following the shallow region of the wetland, there should be a deep pond through which all treated water flows prior to discharge. This region provides an additional opportunity for flow equalization and, if vegetated with floating macrophytes and other deep-water plants, can achieve an additional level of contaminant removal (Dupoldt et al. 1993). In some instances, this terminal deep zone can also be stocked with fish or other aquatic animals for additional biological treatment (Dupoldt et al. 1993).

The height of the outfall from the finishing pond provides depth regulation for the entire wetland and sets the hydraulic grade line for the system (Borin and Tocchetto 2007). This outflow is typically controlled by a weir or flume (Kadlec and Knight 1996), but, if variability in hydraulic grade line is desired, the outlet can be a system in which pipes of various lengths can be inserted to regulate depth (Borin and Tocchetto 2007). The terminal deep zone should be similar in dimensions to the initial sedimentation basin, typically ranging in depth from 2 to 4 m (Dupoldt et al. 1993, Kadlec and Knight 1996). Any outlet configuration selected should be constructed in such a manner as to encourage sheet flow rather than point flow so as to prevent scouring and short-circuiting after the finishing pond (Dupoldt et al. 1993).

#### 4.4.2.1.4 Vegetated spillway

Water from the outlet of the terminal deep pond must be routed from the wetland area back into the main drainage channel. In many wetlands, a vegetated polishing area is constructed after the deep finishing pond in order to provide the final level of treatment and to slow the flow of water prior to discharging into the receiving channel (Dupoldt et al. 1993). This polishing area can also serve as a spillway to connect the outlet of the wetland to the drainage channel. When serving this dual purpose, the polishing area should be bordered by berms in order to guide flow to the drainage channel. The spillway should be vegetated primarily with native grasses that slow the flow of water through the area but are smaller than the macrophyte species that inhabit other regions of the wetland (Millhollon et al. 2009).

#### 4.4.2.2 Vegetation

Plant uptake of nutrients and filtration of microorganisms by vegetation are major removal mechanisms in constructed treatment wetlands. The importance of vegetation in achieving treatment in wetlands has been demonstrated by Huett et al. (2005), Jing et al. (2002), and others. However, those same studies, as well as a number of others including Kadlec and Knight (1996) and Taylor et al. (2006), have demonstrated that the species of vegetation planted within the wetland do not affect the treatment capability of the wetland. Rather than attempting to replicate the planting plan of an existing treatment wetland, it is more important to plant the wetland with a few different species of macrophytes that are either native to the region or well-established in the region and are resilient in wetland conditions (Kadlec and Knight 1996).

Singapore has many native or established species that thrive in wetland conditions (Ong et al. 2012). Some of these species have been observed in other wetland studies to be stable, sustainable, and achieve sufficient contaminant removal, including cattail (*Typha angustifolia*), which was observed by Khatiwada and Polprasert (1999) to achieve high levels of fecal coliform removal and was demonstrated by Ong et al. (2012) to be a reliable plant under wet conditions in Singapore. Other species observed by Ong et al. (2012) to be successful in wetland conditions in Singapore would be good candidates for planting in the shallow vegetated wetland, including dwarf fountain grass (*Pennisetum alopecuroides Hameln*), alligator flag (*Thalia geniculata*), and umbrella sedge (*Cyperus alternifolius*). Species established on berm and basin slopes, as well as the polishing area, should be native grasses rather than the large macrophytes recommended for the shallow wetland (Dupoldt et al. 1993).

Similar to the flexibility in macrophyte species, the location of plants within components of the wetland is not critical to overall wetland effectiveness. While it is important to establish vegetation in the shallow region, spillway, and berm slopes of the wetland as part of construction, plants will likely ultimately become established in locations other than the initial planting arrangement, meaning that initial planting configurations are not critical. This plant re-distribution may represent a more stable state, as plants take hold in regions with conditions favorable to their own needs (Sakadevan and Bavor 1999). Given the evidence that plant species do not significantly alter treatment effectiveness, natural changes to the location of macrophyte species do not need to be inhibited.

The density of vegetation within the shallow wetland region is important for the removal of phosphorus, the critical contaminant in the Neo Tiew subcatchment. According to Kadlec and Knight (1996), plants should be established at a density on the order of 1 kg of biomass per square meter in order to achieve high levels of phosphorus removal. Alternatively, Jordan et al. (2003) report an emergent vegetation cover of 70-90% of the wetland surface in order to achieve high levels of contaminant removal. However, as previously noted, vegetation cover may vary throughout the operational life of the wetland and alter the planting density of any macrophyte species.

In order to achieve consistent treatment performance in a wetland, the vegetation in the wetland must periodically be carefully maintained (Dupoldt et al. 1993). This maintenance should include trimming any grasses in the wetland and removing any dense growth of algae on the water surface, but should not include removal or trimming of any living macrophytes. Though dead plants should certainly be removed before they decompose and contribute nutrients back into the wetland, disturbing living macrophytes may reduce treatment capability by harming the soil

layers of the shallow wetland and inhibiting a plant's ability to transfer oxygen to its roots, where critical treatment processes occur (Dupoldt et al. 1993). Studies have shown that harvesting living biomass is an inefficient and even detrimental method of nutrient removal in wetlands (Kadlec and Knight 1996). Borin and Tocchetto (2007) observed a net removal of nitrogen from a treatment wetland over a five-year period in which established, living macrophytes were undisturbed. Furthermore, disruption of wetland soil may actually release phosphorus, as soil sorption is a major pathway of phosphorus removal (Kadlec and Knight 1996).

#### **4.4.2.3 Other considerations**

##### **4.4.2.3.1 Accessibility**

The accessibility of the wetland for both construction and maintenance purposes is an important design consideration. It should be situated in a location that is accessible by the equipment needed for construction, including earth-moving equipment, and for maintenance (Dupoldt et al. 1993). Additionally, berms and other wetland structures should be constructed in order to facilitate accessibility. External berms should be of sufficient width for a person to walk on them to access different regions of the wetland (Kadlec and Knight 1996). Additionally, the wetland design should incorporate at least two parallel treatment paths in order to increase operational flexibility and facilitate operation of at least part of the wetland during any maintenance activities (Kadlec and Knight 1996).

##### **4.4.2.3.2 Design life**

The expected life of a wetland must be considered when designing the system, both for structural considerations and in order to amortize the initial costs of the wetland over its useful life. Dupoldt et al. (1993) use a 25-year life expectancy for treatment wetland systems in Maine, but report that, with proper design and maintenance, the actual useful life of the system may be considerably longer. Cultural and political conditions must also be considered in the determination of a design life. Given the rapid pace of development in Singapore (Chen et al. 2011), a design life of 25 years may be excessive in this context. Considering the rate of land development in Singapore and the PUB's requirement of a 10-year return period design storm (PUB 2011), a design life of 10 years is suitable for the Neo Tiew subcatchment.

##### **4.4.2.3.3 Soil requirements**

The wetland should be situated in a location that minimizes excavation requirements, particularly changes to the natural topography of the area. Though some excavation and topographic alteration is necessary to match the slope of the wetland to the slope of the drainage channel, the location of the wetland should be selected to as to minimize these requirements and thus minimize both environmental disturbance and construction cost. Excavation should be performed so as to preserve slope stability, and thus altered surfaces should not exceed a slope of 3 horizontal to 1 vertical. Slopes on surfaces within the wetland that are stabilized with vegetation can be as steep as 2 horizontal to 1 vertical (Dupoldt et al. 1993).

The type and depth of soil within and below a wetland are critical to success in sustaining water levels and treatment capability. The topsoil within the wetland must be an adequate substrate for sustaining plant growth and should be at least 0.3 m thick to ensure sufficient depth for plant roots to become established (Sakadevan and Bavor 1999). Below this topsoil layer, there should be a liner to inhibit subsurface flow to or from the wetland. Liners can be a thick (0.4 m or greater) layer of low-permeability clay (Sakadevan and Bavor 1999) or plastic sheeting (Borin



and Tocchetto 2007). The type of liner should be selected based on the local availability of materials, but either type of liner should be installed below the wetland and around its perimeter to restrict not only vertical but also lateral subsurface flow (Borin and Tocchetto 2007). If natural clay underlies the wetland site, a liner is not needed.

#### 4.4.3 Wetland design recommendations

##### 4.4.3.1 Location

The proposed wetland location was selected in order to accommodate the size and accessibility requirements outlined above while still targeting runoff treatment at the most critical location in the drainage channel. As sampling location 3 was observed to have the highest concentrations of most of the contaminants of concern, a location immediately downstream from that location is preferable.

The recommended wetland location is within the sand storage area immediately to the east of Farm 85 Trading and approximately 100 meters downstream from location 3, as can be seen in Figure 36. The intake from the drainage channel to the wetland is located at approximately  $1^{\circ}25'13.875''\text{N}$ ,  $103^{\circ}42'39.4524''\text{E}$  and will enter the wetland at its northeastern corner. The proposed location within the sand storage area meets the size and accessibility requirements for constructing a wetland. Furthermore, the proposed wetland area is currently uncultivated and not utilized for sand storage, meaning that the land could conceivably be converted to wetland without disruption of any current land-use activities.

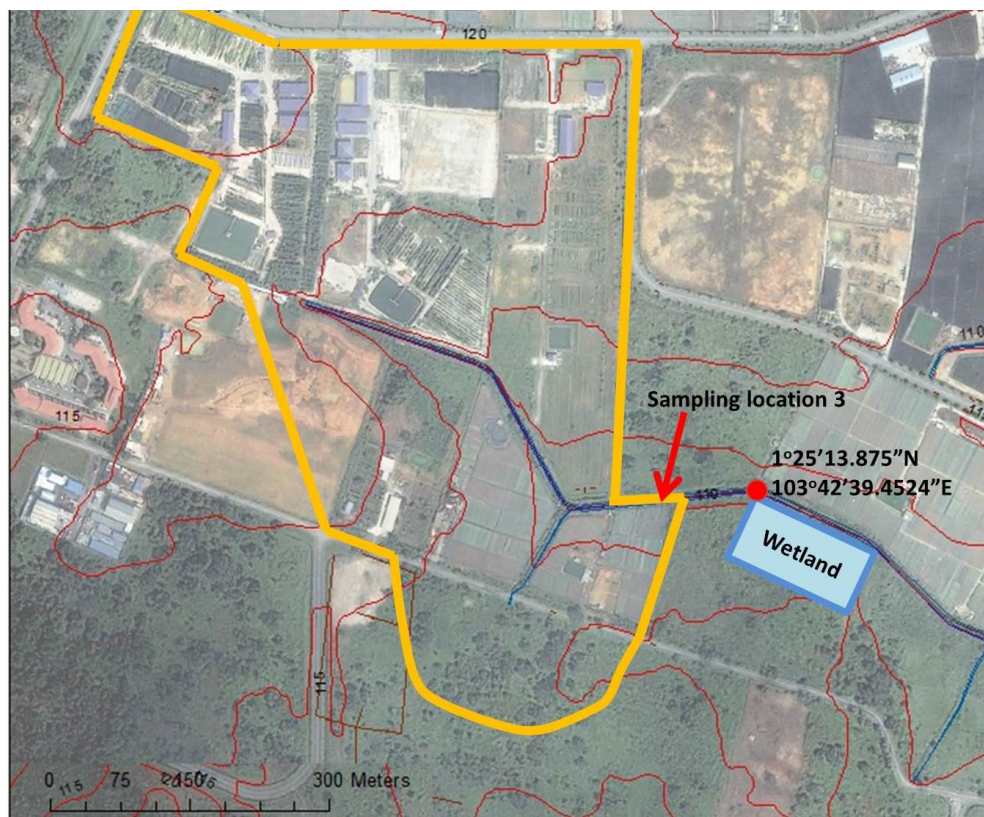


Figure 36: Proposed wetland location.

The proposed location accommodates the required wetland dimensions of 70 m by 140 m, with the longer side of the wetland running parallel to the drainage channel. This location would require alteration of some existing topography in order to match the slope of the wetland to the slope of the drainage channel. Along this reach of the drainage channel, the average slope is 0.5%, with the wetland intake at an elevation of approximately 107.7 m MSL. All elevations included in the wetland design are approximate and should be verified in the field prior to any construction activities.

In order to minimize excavation and earth moving requirements, the wetland should be situated within the existing topography such that the excavated slopes along the west and south sides of the wetland can serve as the external berms of the wetland. In addition to the economic benefits, building into the higher regions adjacent to the wetland minimizes the risk of flooding in surrounding areas.

#### **4.4.3.2 Wetland design**

Plan and profile views of the proposed wetland design are presented in Figures 37 and 38. Detailed design drawings are provided in Appendix D.

Runoff diverted from the drainage channel will first flow into the sedimentation basin. The floor of the sedimentation basin will be excavated to a depth of 2 meters below the surrounding grade in order to facilitate settling of suspended material. The sides of the basin should be graded at a slope of 2 horizontal to 1 vertical and seeded with grass to promote stability (Dupoldt et al. 1993). The sedimentation basin will run the width of the wetland, giving it a length of 68 m. In order to incorporate the required side slopes, the basin should be 10.5 m wide at the top and 3 m wide at the floor.

The outlet of the sedimentation basin to the shallow vegetated wetland will be a series of at least five pipes of small diameter (between 10 and 20 cm) into each shallow wetland cell. This configuration is intended to approximate sheet flow in the water entering the shallow wetland. The outlet pipes will be set into the berm at an elevation of 2.2 m above the floor of the sedimentation basin, or 0.2 m above the surrounding grade. Given these dimensions, the volume of the sedimentation basin is approximately 11,000 m<sup>3</sup>.

There will be two parallel shallow vegetated wetland cells in order to increase operational flexibility. Influent to both of the cells will be introduced through the distributed pipes described above. Each shallow wetland cell will be configured with a sinuous flow channel delineated by internal berms of 0.45 m in height. Each section of the channel is roughly 14-m wide and 32-m long, with a 3-m opening to allow flow into the next section of the channel. In total, there will be eight such channel sections in each wetland cell.

In each wetland cell, two of the channel sections will be transected by deep cross-ditches. The deep cross-ditches will be similar in design to the sedimentation basin, with an excavated floor depth of 2 m and 2 to 1 (horizontal to vertical) side slopes. The cross-ditches will be approximately 10 m long and span the full width of the flow channel (14 m). In each of the two wetland cells, the cumulative volume of the shallow vegetated wetland and deep cross-ditches will be approximately 1,330 m<sup>3</sup>. The final flow channels in each cell will converge at the end of the shallow wetland region and flow directly into the deep finishing pond.

The deep finishing pond will be similar in configuration to the initial sedimentation basin, with an excavated floor depth of 2 m and appropriate side slopes. The finishing pond need not retain water for as long as the sedimentation basin, so the volume can be much less. Thus, the finishing pond will be 25 m long and 10 m wide, with a total volume of approximately 440 m<sup>3</sup>.

Water will exit the deep finishing pond over a broad-crested weir constructed into the external berm surrounding the pond. The weir will be 0.2 m above the surrounding grade (0.15 m below the top of the surrounding berm), with a crest width of approximately 1 m and a total weir length of 7.5 m. Use of a broad-crested weir facilitates sheet flow conditions in the water entering the vegetated spillway (Kadlec and Knight 1996).

Flow exiting the wetland is directed back into the drainage channel through the vegetated spillway. The entrance to the spillway is created by the terminus of the broad-crested weir from the deep finishing pond, which has a width of 7.5 m. The spillway will widen slightly before connecting to the drainage channel. The floor of the spillway should be at the same elevation as the surrounding grade and surrounded by berms in order to direct flow to the drainage channel.

All exterior berms should be sized according to Kadlec and Knight's (1996) instructions, which include consideration of the normal water level, the design storm rainfall, and the loss of freeboard due to sediment and plant accumulation. The designed normal water level for this wetland is 0.45 m, and the design storm rainfall is 32.5 mm (195 mm/hr for 10 minutes). The design life of the wetland is specified above as 10 years, and with a typical freeboard loss rate of 1 cm/yr (Kadlec and Knight 1996), this requires an additional berm height of 10 cm. Thus, exterior berms should be at least 0.58 m in height. External berms should also follow the slope limit of 2 to 1 (horizontal to vertical) and be seeded with grass to aid in slope stability.

In total, the treatment components of the wetland comprise a volume of 4,300 m<sup>3</sup>, which is sufficient to retain the 10-year design storm volume of 4,200 m<sup>3</sup>. The surface area of the treatment wetland, as designed, is approximately 9,800 m<sup>2</sup>, which meets the area requirement of 9,300 m<sup>2</sup> calculated in Section 4.4.1.2.

#### ***4.4.3.3 Connections to drainage channel***

Flow from the drainage channel must be diverted into the wetland by a structure in the drainage channel at the wetland inlet point. Since the wetland is only designed to contain the 10-year design storm, any larger storms could potentially damage the wetland and should not be routed through the treatment wetland. Thus, the structure selected to divert channel flow into the wetland must be capable of diverting flows up to a certain level, but not greater. Flashboards, which are temporary dam structures designed to fail at a specified water level, are an ideal solution for this scenario (USBR 1987).



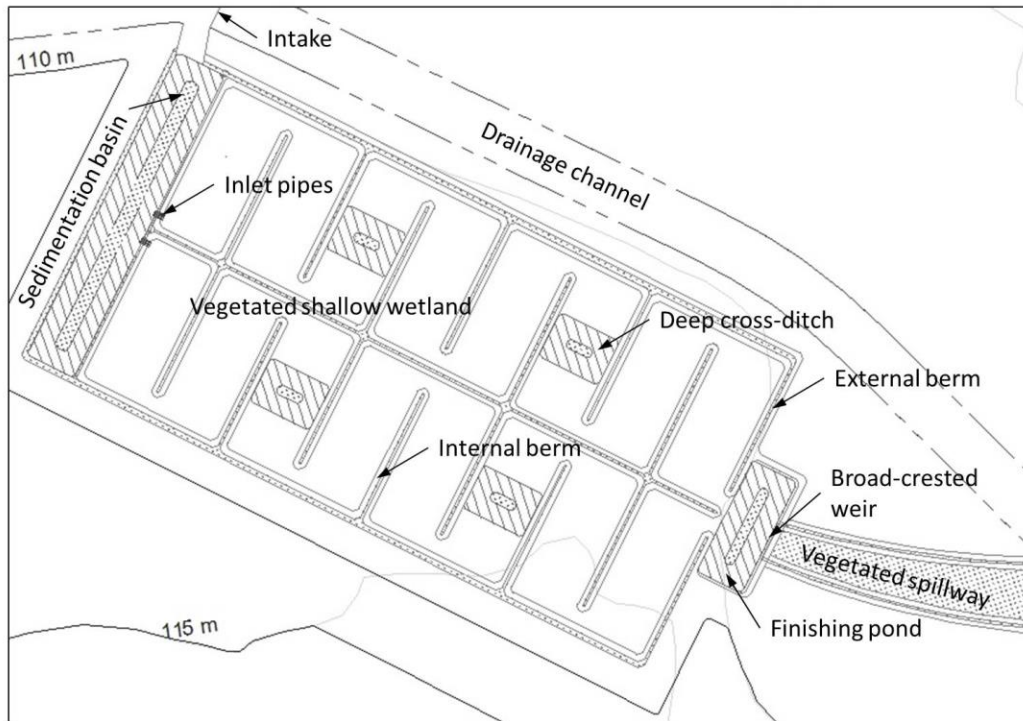


Figure 37: Wetland design, plan view.

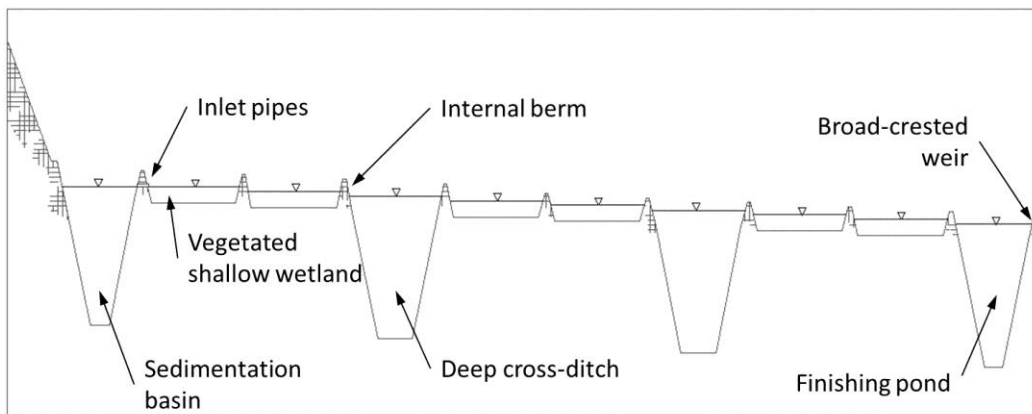


Figure 38: Wetland design, profile view.

Flashboards can exist in a number of configurations, including those that must be removed prior to large rainfall events and those intended to fail at a specified level as described above (Linsley and Franzini 1972). As removal of a structure before a storm event requires fairly accurate rainfall forecasting and is operationally intensive, flashboards designed to fail above the 10-year design storm level are preferable for this application. However, this configuration has pitfalls, both economic and operational. If the designed failure point is reached frequently, it can be expensive to continually replace the flashboards. Additionally, though the system is designed to a specific failure point, the precise moment of failure is uncertain, and, when the dam does fail, a large volume of water can be released instantaneously (USBR 1987). In applications with downstream areas sensitive for human or ecological safety, flashboards designed for failure may

not be appropriate. However, for the Neo Tiew subcatchment, with downstream land uses including industrial storage and agriculture, flashboards should be designed to fail above the 10-year design storm water level.

First, the water level in the channel for the 10-year design storm must be determined. This can be done using Manning's equation, as seen in Equation (4) (PUB 2011).

$$Q_c = \frac{1}{n} A R_h^{2/3} S^{1/2} \quad (4)$$

In which:

- $Q_c$  = flow through drain ( $\text{m}^3/\text{s}$ )
- $n$  = roughness coefficient, 0.015 for a concrete channel (PUB 2011)
- $A$  = flow area ( $\text{m}^2$ ), dependent on height of water surface above bed
- $R_h$  = hydraulic radius (m), dependent on height of water surface above bed
- $S$  = bed gradient, 0.005 for this reach of the channel

Setting  $Q_c$  equal to the runoff flow resulting from the 10-year design storm ( $7 \text{ m}^3/\text{s}$ ) and applying the equation to the channel cross-section geometry, as shown in Figure 39, results in a water level of 0.96 m for the 10-year design storm.

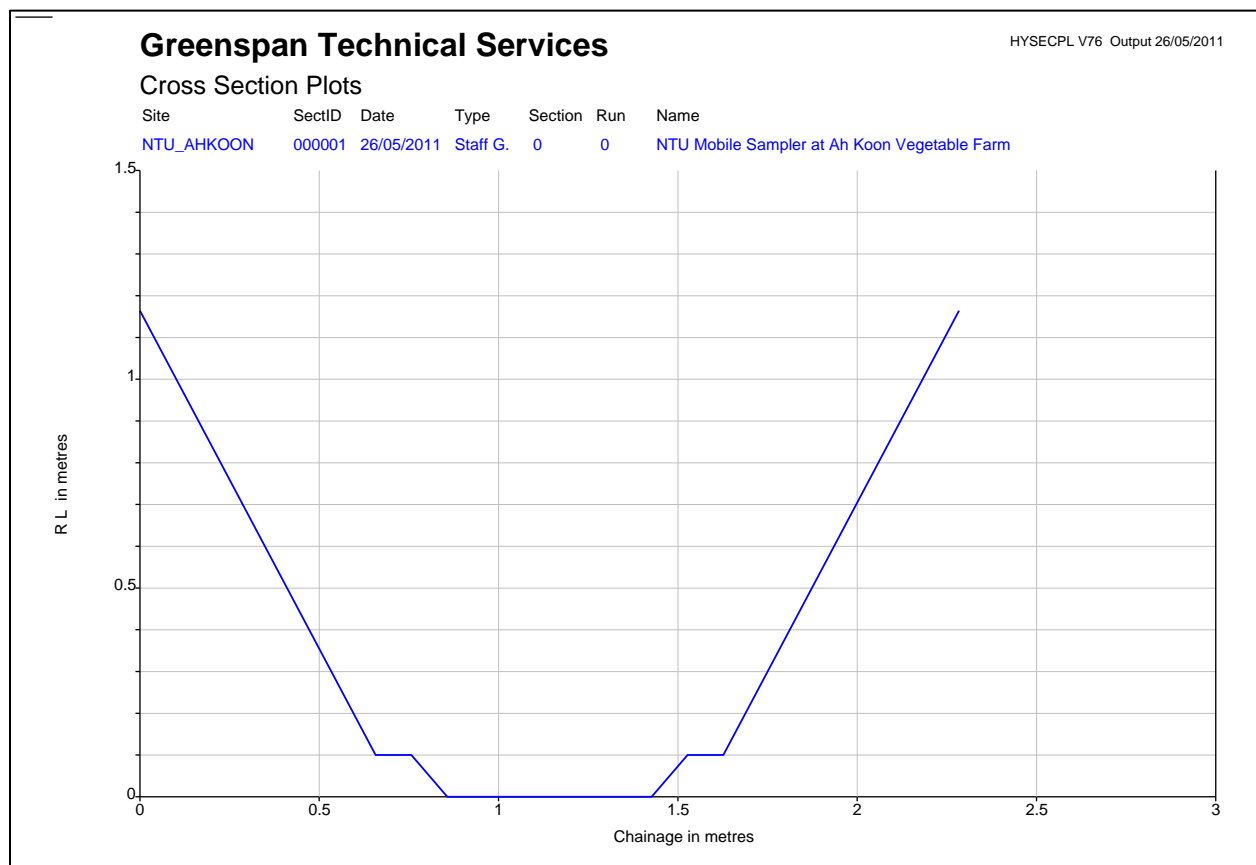
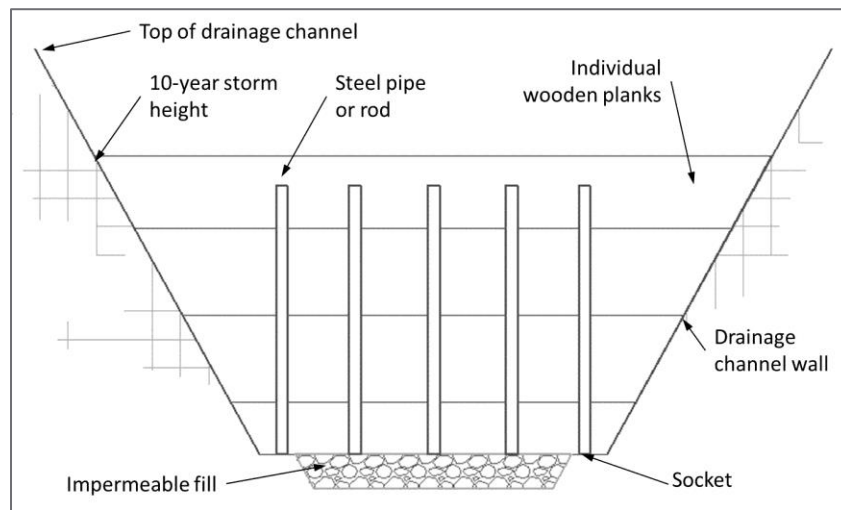


Figure 39: Drainage channel cross-section (Le 2013).

The height of the flashboards in the channel should be equal to the water level for the 10-year design storm, 0.96 m. This allows for a freeboard height of approximately 0.3 m in the drainage channel. The flashboard structure should consist of a number of individual wooden planks or panels laid horizontally across the drainage channel (USDOI 1987) and anchored to the channel by number of vertical pins (Linsley and Franzini 1972). The support pins are typically “steel pipe or rod set loosely in sockets in the [floor of the channel] and designed to bend and release the flashboards at the desired water level” (Linsley and Franzini 1972). This configuration can be seen in Figure 40, a schematic diagram of the downstream side of the flashboard structure. The bottom shelf of the channel will be filled with a fixed, impermeable material, such as concrete, to ensure that the lowest level of flow in the channel is always diverted into the wetland, even when the flashboards fail. The flashboard structure will thus begin at the bottom of the larger trapezoidal portion of the channel cross-section, as shown in Figure 40.



**Figure 40: Drainage channel flashboard design, downstream side.**

The connection to the channel from the vegetated spillway on the downstream side of the wetland need not be as rigorously designed as the inlet structure. Water flowing in the vegetated spillway will resemble distributed, sheet flow, and the outlet structure to the drainage channel can maintain this with a concrete spillway entrance, a structure that is already in common use throughout the subcatchment.

#### **4.4.4 Projected performance**

Since the wetland is designed to meet design-storm requirements rather than specific treatment performance objectives, it is important to determine what the treatment capability of the wetland will be if implemented as designed. Treatment performance is difficult to determine accurately, as the level of treatment depends greatly upon the residence time of a given parcel of water within the wetland (Kadlec and Knight 1996), which in turn depends on the irrigation and precipitation flows into the wetland and the residence time distribution of the wetland. Contaminant removal capacity of a wetland is thus perhaps best approximated on an annual basis, which averages the widely variable treatment range over a long time period.

#### 4.4.4.1 Nutrients

Nutrient removal in wetlands is often represented with area-based kinetics (Kadlec and Knight 1996), as shown in Equation (5).

$$J = k_A(c - c^*) \quad (5)$$

In which:

- $J$  = annual contaminant reduction rate ( $\text{g/m}^2/\text{yr}$ )
- $k_A$  = removal rate constant ( $\text{m/yr}$ )
- $c$  = influent contaminant concentration ( $\text{g/m}^3$ )
- $c^*$  = background contaminant concentration ( $\text{g/m}^3$ )

The annual contaminant mass removal can be translated to an annually-averaged contaminant concentration reduction by first multiplying by the wetland area ( $9,800 \text{ m}^2$ ) and then dividing by the annual volume of flow through the wetland. This average reduction can then be subtracted from the observed contaminant concentrations to predict average effluent water quality from the wetland. Background pollutant concentrations depend on climate, vegetation, and other highly variable factors that have not been determined for the Neo Tiew subcatchment and are typically small relative to influent concentrations (Kadlec and Knight 1996), so this term will be neglected for the purposes of this discussion.

This performance prediction method relies heavily upon average values of both the contaminant removal rate and wetland flow, and thus determination of these average values must be made carefully. Removal rates for each contaminant of concern can be approximated based on studies of similar treatment wetlands, but since accurate determination of this parameter for the Neo Tiew wetland is difficult, a representative range of values from the literature is used in this study (Kadlec and Knight 1996). The average flow conditions through the wetland can be approximated from the channel flow observed during field work. Since flow rate was not observed at sampling location 3, the best approximation of flow at this location can be made by scaling up the flow observed at sampling location 2 in proportion to the additional area that drains to location 3 relative to location 2. As treatment performance is important in both wet and dry weather, both flow rates are used to project contaminant removal capability under different conditions.

Using the same subcatchment delineation method outlined in Section 4.4.1.2.2, the subcatchment area draining to sampling location 2 is approximately 13.2 ha. As determined in Section 4.4.1.2.2, the area draining to sampling location 3 is approximately 28 ha. Thus, the scaling factor for flow observed at location 2 to flow at location 3 should be 2.1. The wet-weather flow observed at sampling location 2 on 1/22/13 was  $350 \text{ m}^3/\text{d}$ , resulting in a linearly-scaled average flow at location 3 of  $735 \text{ m}^3/\text{d}$ . This represents an annualized flow of  $268,000 \text{ m}^3/\text{yr}$ . The dry-weather flow observed at sampling location 2 on 1/25/13 was  $150 \text{ m}^3/\text{d}$ , giving an average flow at location 3 of  $315 \text{ m}^3/\text{d}$  and annualized flow of  $115,000 \text{ m}^3/\text{yr}$ .

##### 4.4.4.1.1 Phosphorus

In FWS wetlands with similar hydraulic loading rates and in similar climates to the designed Neo Tiew wetland, total phosphorus removal rates (parameter  $k_A$  in Equation (5)) ranging from 3 to 21  $\text{m/yr}$  have been observed (Kadlec and Knight 1996). The average total phosphorus

concentration observed at sampling location 3 was 1.9 mg/L. Using these values with Equation (5), an average annual removal of between 63 and 390 kg of total phosphorus is predicted. Using the annualized wet-weather flows above, this represents an average TP concentration reduction of between 0.24 and 1.5 mg/L. Again using the average influent concentration of 1.9 mg/L, this predicts an average effluent TP concentration of between 0.4 and 1.66 mg/L. This represents a 13 to 79% reduction in TP concentration in the wetland. Using the dry-weather flow rate observed, the predicted effluent concentration ranges from 0 to 1.35 mg/L, representing a TP reduction between 29 and >99%.

#### 4.4.4.1.2 Nitrogen

Observed removal rates for total nitrogen in wetlands similar to the Neo Tiew wetland range from 14 to 22 m/yr (Kadlec and Knight 1996). The average observed total nitrogen at sampling location 3 was 19 mg/L. Using the same treatment prediction calculations as performed for total phosphorus, under the observed wet-weather conditions, the wetland would remove between 2,600 and 4,100 kg of total nitrogen per year, which equates to a predicted TN concentration reduction of between 9.7 and 15.3 mg/L. With an average influent of 19 mg/L, this results in an effluent TN concentration of between 3.7 and 9.3 mg/L, or a 51 to 80% reduction in TN concentration. Under the observed dry-weather conditions, the wetland is predicted to remove over 99% of the influent total nitrogen.

#### 4.4.4.2 Bacteria

Bacterial removal in constructed treatment wetlands has been observed to follow first-order reaction kinetics (Kadlec and Knight 1996, Khatiwada and Polprasert 1999). The effluent bacterial count from a treatment wetland is thus predicted as shown in Equation (6) (Khatiwada and Polprasert 1999).

$$c = c_o e^{-k\tau} \quad (6)$$

In which:

- $c$  = effluent bacterial count (MPN/100 mL)
- $c_o$  = influent bacterial count (MPN/100 mL)
- $k$  = removal rate constant ( $\text{days}^{-1}$ )
- $\tau$  = hydraulic residence time, HRT (days)

For constructed treatment wetlands in a laboratory setting with a hydraulic residence time of 5.5 days, Khatiwada and Polprasert (1999) report a fecal coliform removal rate constant of  $1.63 \text{ days}^{-1}$ . The mean hydraulic residence time of the Neo Tiew wetland can be calculated by dividing the design volume by the mean daily flow rate under wet- and dry-weather conditions. This provides a mean HRT of 5.8 days under the observed wet-weather conditions and 13.6 days for dry-weather flows. Given the similarity of the mean wet-weather HRT to that of the wetland studied by Khatiwada and Polprasert (1999), it seems that the removal rate constant determined by their study can reasonably be applied to the designed wetland.

Khatiwada and Polprasert's (1999) study focused specifically on fecal coliform, and thus the bacterial counts used to predict wetland treatment performance should be those for *E. coli*, which is of fecal origin. The average observed *E. coli* count at sampling location 3 was 38,000 MPN/100 mL. Using a removal rate constant of  $1.63 \text{ days}^{-1}$ , the predicted effluent *E. coli* count

from the wetland is 3 MPN/100 mL during wet-weather flow and less than 1 MPN/100 mL in dry-weather flow. In both average wet- and dry-weather conditions, the wetland is predicted to achieve greater than 99% removal of fecal coliform.

#### **4.4.4.3 Solids**

Solids removal performance is not typically calculated using kinetics as the removal is largely due to the physical process of sedimentation, which occurs predominantly in the first deep region of the wetland (Kadlec and Knight 1996). The median TSS removal in FWS constructed wetlands is approximately 75% (Kadlec and Knight 1996). Using the observed average TSS concentration at sampling location 3 of 32.8 mg/L, this predicts a wetland effluent TSS concentration of 8.2 mg/L.

#### **4.4.4.4 Discussion**

The predicted treatment performance of the designed wetland is summarized in Table 8. It is evident from Table 8 that there is a range of treatment capability across different contaminants within the wetland. Furthermore, there are broad ranges of treatment performance predictions for each contaminant based on the choice of removal rate constant and runoff flow rate. Unfortunately, contaminant removal rates for the wetland cannot be accurately predicted prior to construction of the wetland due to the site-specific nature of treatment. However, more flow data for the subcatchment would help to narrow the predicted treatment range by establishing a more accurate picture of “normal” flow conditions for the area. Similarly, the average influent contaminant concentration values are based on only two sampling events. More sampling must be conducted in order to establish truly representative average influent concentrations. Another potential source of error in the performance calculations is the neglect of the background concentration term,  $c^*$ . Though this term is usually small (typically less than 1.5 mg/L), it may affect the predicted treatment performance (Kadlec and Knight 1996). Lastly, it must be remembered that the treatment performance of a wetland changes over time. In particular, the first year of operation of a treatment wetland does not achieve the same treatment performance as subsequent years, as vegetation and other wetland features have not yet stabilized (Sakadevan and Bavor 1999). Thus, the projected performance values calculated above reflect an average over the life of the wetland, rather than a specific value at a given point in time.

In general, the broad range of predicted treatment performance is consistent with observed treatment performance presented in the literature. As noted in Section 4.2, previously-studied constructed wetlands have exhibited wide variation in treatment ability, ranging from anywhere between 20 to greater than 90% removal of nutrients. Such variability is not only observed in comparisons of different wetlands, but can also be exhibited within the same wetland over different seasons or influent characteristics.

Though the reduction percentages presented in Table 8 represent a wide range of predictions, it must be remembered that the receiving body of water, Kranji Reservoir, is sensitive to nutrient inputs, especially phosphorus, and any reduction achieved is worthwhile. Furthermore, given the first-order removal kinetics observed in wetland treatment (Kadlec and Knight 1996), the wetland’s treatment performance will be highest when the influent contaminant concentration is highest. The wetland will be both most necessary and most effective in the critical situations during which there is very high contaminant concentrations in runoff.



Table 8: Projected average wetland treatment performance.

Contaminant, weather conditions	Removal rate (m/yr)	Influent concentration (mg/L)	Effluent concentration (mg/L)	Percent reduction (%)
Total phosphorus - wet	3 - 21	1.9	0.4 - 1.7	13 - 79
Total phosphorus - dry	3 - 21	1.9	0 - 1.4	29 - >99
Total nitrogen - wet	14 - 22	19	3.7 - 9.3	51 - 80
Total nitrogen - dry	14 - 22	19	0	>99
<i>E. coli</i> - wet <sup>a</sup>	1.63 <sup>b</sup>	38,000	3	>99
<i>E. coli</i> - dry <sup>a</sup>	1.63 <sup>b</sup>	38,000	<1	>99
Total suspended solids	n/a	33	8.2	75

a: *E. coli* concentrations reported in MPN/100 mL

b: *E. coli* removal rate in units of day<sup>-1</sup>

#### 4.5 Non-structural BMP recommendations

In addition to the construction of a structural best management practice for runoff treatment, there are several promising agricultural management methods for upstream mitigation of contaminant runoff. The practice of cyclic irrigation, in which crops are watered in a series of short irrigation applications throughout the day, has been shown to reduce both the volume of and contaminant concentrations in plant nursery runoff (Bilderback 2002). This practice could easily be adapted to both plant nurseries and other agricultural operations in the Neo Tiew subcatchment. The intensive vegetable cultivation at Farm 85 Trading is similar in scale and irrigation scheme to the systems in operation at many containerized plant nurseries and would thus likely reap the same environmental benefits from cyclic irrigation. This practice does not measurably reduce plant growth and has no implementation cost other than slightly increased operational intensiveness. Furthermore, by reducing the water withdrawals of a farm, cyclic irrigation can actually result in savings to the farm operator (Bilderback 2002).

Though specific fertilization quantities and application practices of the properties within the Neo Tiew subcatchment are unknown, it is evident from site observations that many of the operations in the area apply fertilizer to crops. One option to mitigate nutrient and bacterial contamination in runoff, while still enabling high rates of plant growth, is to simply rearrange crops based on fertilizer requirements. The most nutrient-intensive plants should be located furthest away from the drainage channel, with the intent of allowing for more infiltration of runoff from these fertilizer-laden crops prior to reaching the drainage outlet (EPA 1992). Additionally, the actual fertilizer needs of each crop should be evaluated so as to not over-fertilize, which is both an environmental and economic disadvantage (EPA 1992).

Lastly, given the rapid development observed and projected to continue in Singapore (Chen et al. 2011), it is possible that the Neo Tiew subcatchment may not be dominated by agricultural land use in the future. Modeling of nonpoint source pollution indicates that urbanization reduces the concentration of contaminants in surface runoff due to the changes in nutrient and bacteria application that come with a change in land use. Specifically, less fertilizer is applied to open land in urban areas than in agricultural areas (Tsihrantzis et al. 1996). Though a discussion of the overall ecological merits or faults of urbanization is outside the scope of this study, the development of the subcatchment would likely alleviate some of the nutrient and bacteria loading to Kranji Reservoir.



## 5. Conclusions and Recommendations

### 5.1 Field work

The field- and laboratory work conducted in January 2013 sought to determine the primary source and degree of agricultural nonpoint source pollution in the Neo Tiew subcatchment of Kranji catchment in Singapore. Grab water quality samples were collected over two days in wet- and dry-weather conditions along the length of the main drainage channel that runs throughout the subcatchment. Samples were analyzed for nitrogen, phosphorus, and bacterial contamination, as well as total suspended solids.

It is evident from the results of the sampling and analysis program that there is extensive nonpoint source nutrient and bacterial pollution in the subcatchment, with a maximum observed total nitrogen concentration of 19.8 mg/L and a maximum total phosphorus concentration of 2.12 mg/L. Observed bacterial counts were also high, with a maximum observed total coliform count of 1,050,000 MPN/100 mL, a peak *E. coli* count of 166,000 MPN/100 mL, and a peak enterococci count of 50,400 MPN/100 mL. These high runoff nutrient concentrations and bacteria counts are likely a result of fertilizer application at agricultural operations within the subcatchment.

The peak concentrations observed for the majority of the contaminants occurred at sampling location 3, which is immediately downstream from Farm 85 Trading, an intensive row-cropped vegetable farm. Though high contaminant concentrations were also observed at locations upstream of Farm 85 Trading, it is evident from the collected samples that Farm 85 Trading is a primary source of nonpoint source pollution in the subcatchment.

An effort was also made to determine average flow conditions in the drainage channel under wet- and dry-weather conditions. Channel dimensions were taken at points of supercritical flow and used to calculate flow rates. At the sampling location immediately upstream of Farm 85 Trading, observed wet-weather flow was 350 m<sup>3</sup>/d and observed dry-weather flow was 150 m<sup>3</sup>/d. While the measurements and samples taken provide insight into flow conditions and contaminant concentrations on the given sampling days, additional sampling is needed in order to understand truly representative water quality and quantity conditions.

### 5.2 Pollution control

Upon determination of the nature and source of the nonpoint source pollution in the Neo Tiew subcatchment, recommendations were made for best management practices to limit the introduction of this pollution into Kranji Reservoir. Best management practice recommendations include both a structural system to treat polluted runoff and an upstream management plan to limit the introductions of contaminants into agricultural runoff.

A constructed treatment wetland is the most appropriate BMP for mitigating the effects of agricultural runoff in this application. Constructed wetlands are able to treat a broad range of contaminants and accommodate highly variable flow rates (Sharma et al. 2008), both of which may be expected of the runoff from the Neo Tiew area. The proposed treatment wetland is designed to accommodate the 10-year design storm in accordance with PUB guidelines (PUB 2011). The wetland consists of a deep sedimentation forebay, two parallel shallow vegetated wetland cells with deep transverse ditches, a deep finishing pond, and a vegetated spillway. The

designed wetland occupies an area of 0.98 ha and has a volume of 4,300 m<sup>3</sup>. In order to address the pollution in the location of most immediate concern, the wetland has been sited immediately downstream of sampling location 3, where peak contaminant concentrations were observed.

The projected wetland treatment performance can be determined using the observed contaminant concentrations and flow rates in conjunction with established kinetic removal rate constants (Kadlec and Knight 1996). As designed, the wetland could remove between 13 and 99% of influent total phosphorus, 51 to 99% of influent total nitrogen, greater than 99% of influent fecal coliform, and approximately 75% of influent total suspended solids. The uncertainty in these performance calculations is due to variability in average runoff flow rates and uncertainty in site-specific wetland removal rate constants.

In addition to treating contaminated runoff downstream from agricultural operations, upstream best management practices should also be implemented to minimize the introduction of contaminants to surface runoff. Two management practices that have been shown to be effective and economically feasible are cyclic irrigation and crop rearrangement. Cyclic irrigation utilizes existing irrigation equipment to deliver hydration to plants in short doses throughout the day, minimizing irrigation volume, and thus runoff quantity, while still enabling achievement of full plant growth capabilities (Bilderback 2002). Crop rearrangement places the most fertilizer-intensive species at locations further from receiving water bodies with the intent of facilitating infiltration of pollutant-laden runoff prior to reaching the drainage channel (EPA 1992).

### **5.3 Recommendations and need for additional work**

Nonpoint source pollution is an increasingly important problem in agricultural watersheds around the world (Easton et al. 2009) and Singapore's Kranji catchment is no exception. It is clear from this study, as well as those of Le (2013), Bossis (2011), and Dixon et al. (2009), that there is significant nutrient and bacterial pollution of surface runoff in the catchment in general and in the Neo Tiew subcatchment in particular. There are a number of best management practices recommended in this study for the alleviation of nonpoint source pollution, all of which aim to mitigate the detrimental effects of pollution on Kranji Reservoir.

Though this study determined the primary source of agricultural nonpoint source pollution in the Neo Tiew subcatchment, additional work is needed to solidify the understanding of site conditions and to enable more precise predictions of wetland performance. A more extensive sampling regimen is necessary to establish what constitutes normal water quality and quantity conditions in the area. Additional sampling days spread over a broader range of weather and seasonal conditions would help to achieve this goal. More accurate estimates of average influent flow rates and contaminant concentrations would facilitate a better prediction of wetland treatment performance. Additionally, there are some critical wetland parameters that cannot be determined prior to construction, including residence time distribution and contaminant removal rate constants. Both of these parameters provide important insight into the operation and performance of the constructed wetland. Though a comprehensive understanding of the subcatchment is important for design and construction of the wetland, additional study is necessary after implementation of the treatment wetland.

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## Appendix A: Sampling location information

Table A-1: Sampling location information.

Sample ID	Latitude	Longitude	Description
1	N01,25',20.6"	E103,42',23.5"	Mao Sheng QuANJI Landscaping outfall (start of channel)
2	N01,25'17.4"	E103,42',30.2"	Upstream from Farm 85 Trading (at boundary between Farm 85 Trading and Green Circle Farm)
3	N01,25',13.0"	E103,42',36.7"	Downstream from Farm 85 Trading (near sand storage boundary fence)
4	N01,25',04.3"	E103,42',57.1"	Upstream from bridge over Neo Tiew Rd. (between sand storage and bridge)
5	N01,25',04.4"	E103,42',57.7"	Downstream side of bridge over Neo Tiew Rd. (between road bridge and Bollywood bridge)
6	N01,25',03.2"	E103,43',03.6"	Downstream side of Bollywood pedestrian bridge
7	BLANK	BLANK	Field blank (DI water collected in same glassware/Whirl-Paks as field samples. Traveled with field samples.)
8	N01,25',14.6"	E103,42',31.9"	Parallel drainage ditch in Farm 85 Trading
9	N01,25'15.0"	E103,42'28.4"	Pond at Green Circle farm. Sample taken near surface on western edge of pond.

## Appendix B: Channel flow measurements

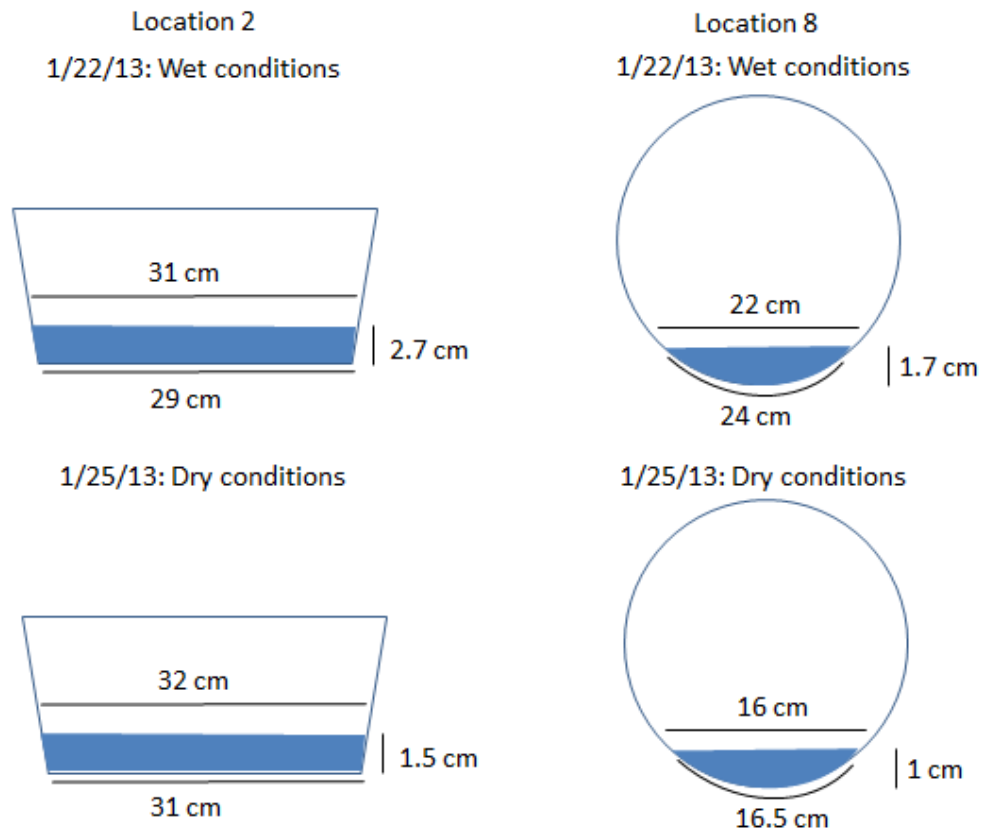


Figure B-1: Measured channel dimensions.

## Appendix C: Measured contaminant concentrations

Table C-1: Measured contaminant concentrations.

Sample ID	1-1	2-1	3-1	4-1	5-1	6-1	7-1	8-1
Date	1/16/2013	1/16/2013	1/16/2013	1/16/2013	1/16/2013	1/16/2013	1/16/2013	1/16/2013
Time	10:11 AM	10:34 AM	11:00 AM	11:38 AM	11:30 AM	11:49 AM	9:00 AM	10:39 AM
Conditions	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry
NH-N (ammonia) ppm	0.35	2.14	0.17	0.04	0.17	0.35	0.03	0.04
NO-N (nitrate) ppm	0.3	0.98	16.8	6.16	5.84	5.74	0.08	78.8
NO-N (nitrite) ppm	0.08	0.13	0.21	0.23	0.18	0.18	0.01	0.23
TN (total nitrogen) ppm	1.8	4.5	19.8	6.57	6.2	6.09	<0.01	81.3
TDP (dissolved) ppm	0.32	0.16	1.27	0.38	0.29	0.4	0.0081	0.37
TOP (organic) ppm	0.052	0.034	1.04	0.13	0.3	0.14	0.09	0.066
TP (total phosphate) ppm	0.52	1.37	2.12	0.66	0.79	0.76	0.81	0.45
TSS ppm	10.4	2	37.2	8.4	7.6	4.8	<2	3.6
TC MPN/100 mL	697,000	155,310	1,046,200	290,900	272,300	325,500	<1	195,600
E. Coli MPN/100 mL	11,690	45,690	12,033	934	1,002	1,039	<1	985
Enterococci MPN/100 mL	2,160	11,620	7,400	2,105	3,654	1,785	<1	4,611

Table continued on next page.

Table C-1: Continued.

Sample ID	1-2	2-2	2-2D	3-2	4-2	5-2
Date	1/22/2013	1/22/2013	1/22/2013	1/22/2013	1/22/2013	1/22/2013
Time	9:57 AM	10:25 AM	10:25 AM	10:55 AM	11:56 AM	11:36 AM
Conditions	Wet	Wet	Wet	Wet	Wet	Wet
NH-N (ammonia) ppm	1.18	0.83	2.66	1.41	0.62	0.66
NO-N (nitrate) ppm	2.33	1.08	1.05	17	6.71	6.37
NO-N (nitrite) ppm	0.09	0.15	0.15	0.24	0.21	0.22
TN (total nitrogen) ppm	4.03	5.34	n/a	18.1	6.98	6.65
TDP (dissolved) ppm	0.57	0.12	n/a	1.2	0.31	0.32
TOP (organic) ppm	0.07	0.006	n/a	0.22	0.059	0.077
TP (total phosphate) ppm	0.68	0.24	0.29	1.6	0.49	0.54
TSS ppm	10	2.4	n/a	28.4	2.8	2.6
TC MPN/100 mL	684,800	325,500	325,500	686,700	124,600	185,000
E. Coli MPN/100 mL	49,600	166,400	142,100	64,400	2,187	2,489
Enterococci MPN/100 mL	15,531	49,600	51,200	27,200	7,701	4,352

Table continued on next page.

Table C-1: Continued.

Sample ID	6-2	7-2	8-2	9-2
Date	1/22/2013	1/22/2013	1/22/2013	1/22/2013
Time	11:46 AM	9:00 AM	10:40 AM	12:15 PM
Conditions	Wet	Wet	Wet	Wet
NH-N (ammonia) ppm	0.52	<0.01	0.01	2.77
NO-N (nitrate) ppm	3.4	0.03	53.2	0.08
NO-N (nitrite) ppm	0.21	0.01	0.2	0.05
TN (total nitrogen) ppm	7.46	<0.01	55	3.77
TDP (dissolved) ppm	0.32	0.015	0.3	2.07
TOP (organic) ppm	0.053	0.099	0.009	0.19
TP (total phosphate) ppm	0.64	0.08	0.44	1.43
TSS ppm	3.4	<2	7	10.4
TC MPN/100 mL	142,100	<1	209,800	3,090,000
E. Coli MPN/100 mL	1,334	<1	1,515	85,700
Enterococci MPN/100 mL	3,654	<1	9,800	142,100

## Appendix D: Wetland design drawings

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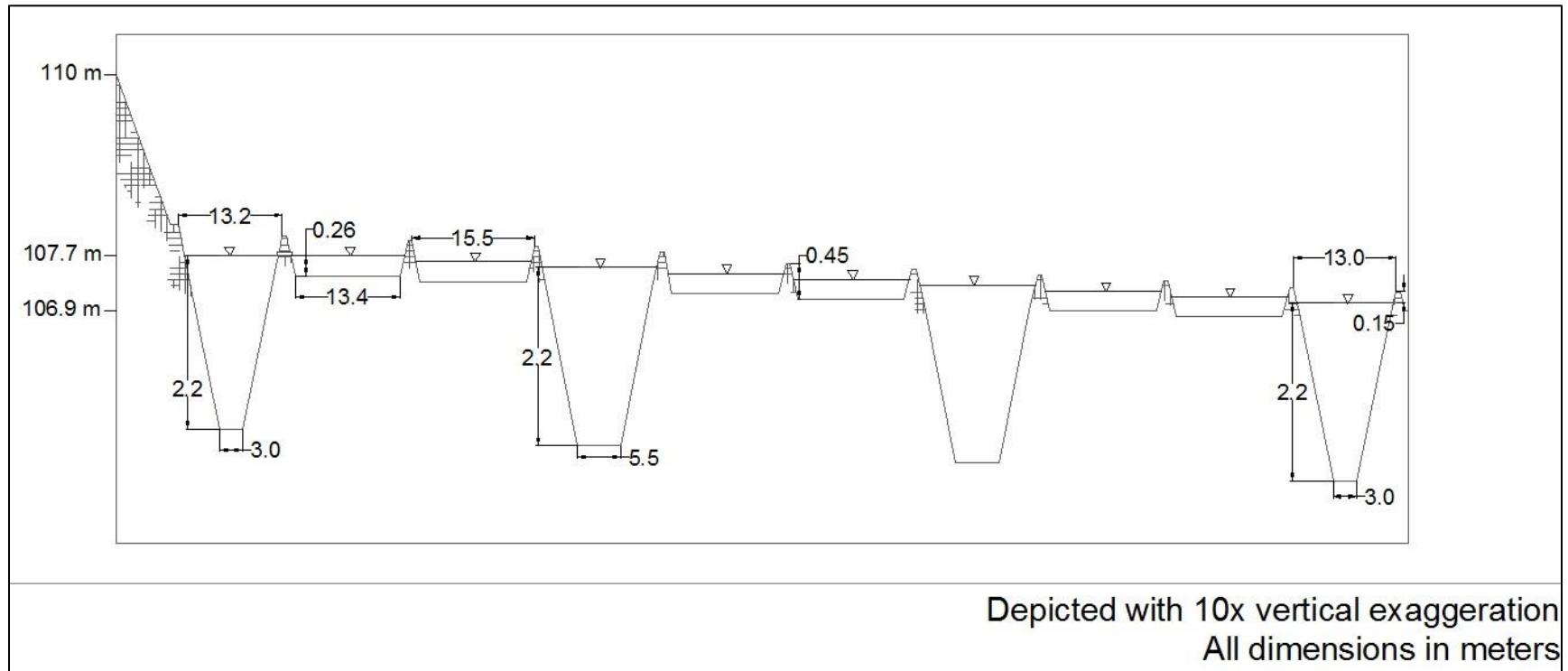


Figure D-1: Wetland design with dimensions, profile view.



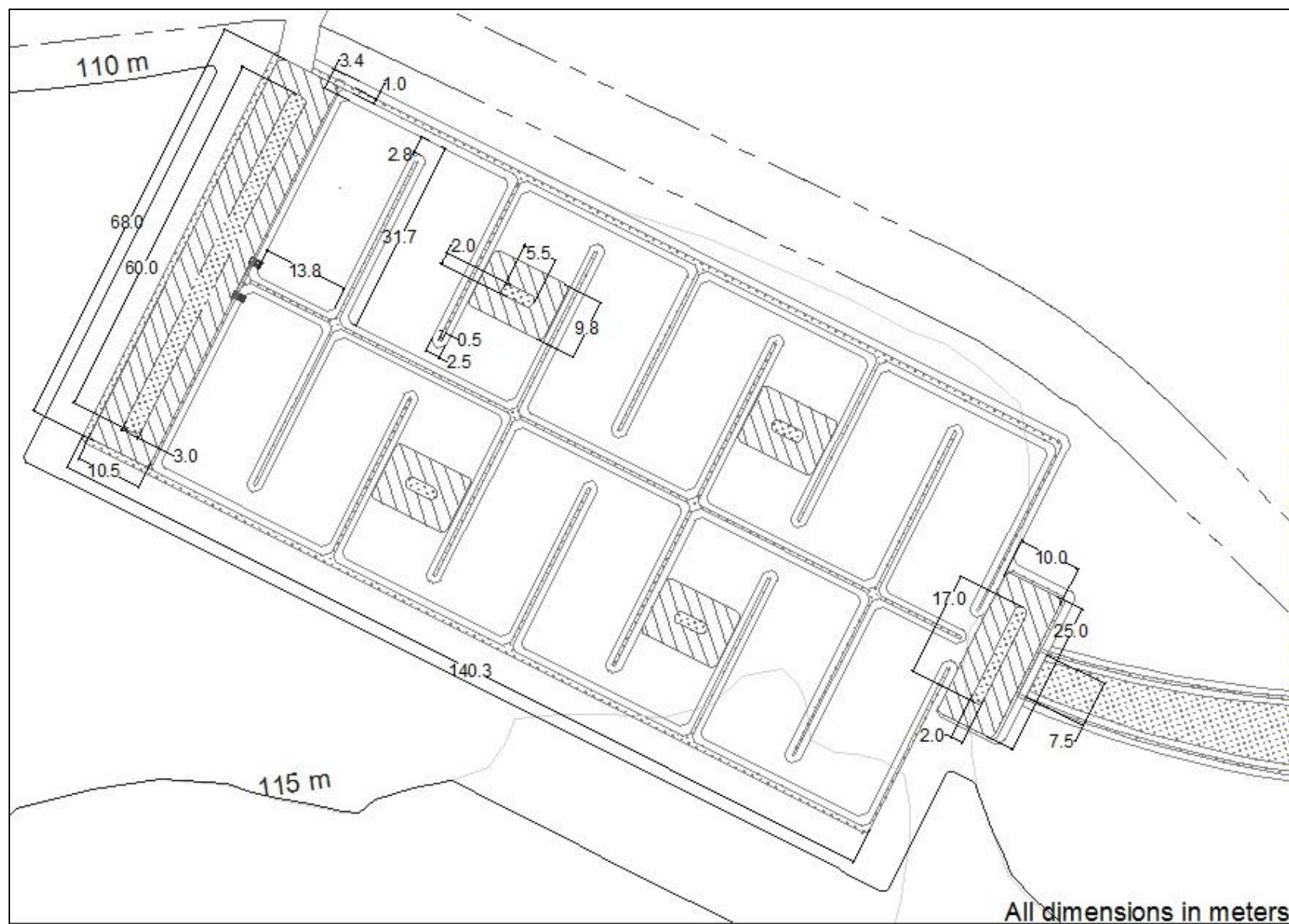


Figure D-2: Wetland design with dimensions, plan view.

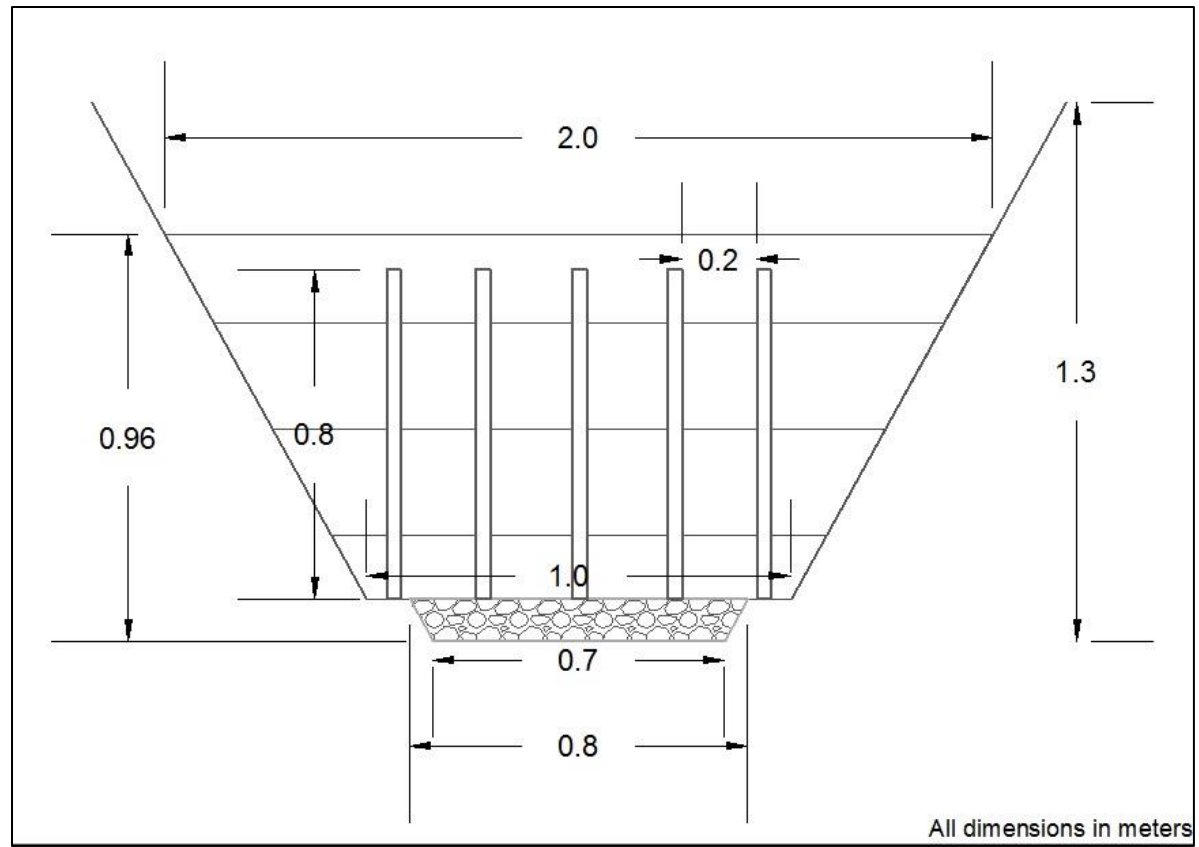


Figure D-3: Channel flashboard design with dimensions, downstream view.